Land Change Science

Observing, Monitoring and Understanding Trajectories of Change on the Earth’s Surface

Edited by
Garik Gutman,
Anthony C. Janetos,
Christopher O. Justice,
Emilio F. Moran,
John F. Mustard,
Ronald R. Rindfuss,
David Skole,
Billy Lee Turner II and
Mark A. Cochrane

Springer
Remote Sensing and Digital Image Processing

VOLUME 6

Series Editor:

Freek D. van der Meer, Department of Earth Systems Analysis, International Institute for Geo-Information Science and Earth Observation (ITC), Enschede, The Netherlands & Department of Geotechnology, Faculty of Civil Engineering and Geosciences, Technical University Delft, The Netherlands

Editorial Advisory Board:

Michael Abrams, NASA Jet Propulsion Laboratory, Pasadena, CA, U.S.A.
Paul Curran, Department of Geography, University of Southampton, U.K.
Arnold Dekker, CSIRO, Land and Water Division, Canberra, Australia
Steven M. de Jong, Department of Physical Geography, Faculty of Geosciences, Utrecht University, The Netherlands
Michael Schaepman, Centre for Geo-Information, Wageningen UR, The Netherlands

The titles published in this series are listed at the end of this volume.
LAND CHANGE SCIENCE

Observing, Monitoring and Understanding Trajectories of Change on the Earth’s Surface

Edited by

GARIK GUTMAN
NASA Headquarters, Washington, DC, U.S.A.

ANTHONY C. JANETOS

CHRISTOPHER O. JUSTICE
University of Maryland, College Park, MD, U.S.A.

EMILIO F. MORAN
Indiana University, Bloomington, IN, U.S.A.

JOHN F. MUSTARD
Brown University, Providence, RI, U.S.A.

RONALD R. RINDFUSS
University of North Carolina, Chapel Hill, NC, U.S.A.

DAVID SKOLE
Michigan State University, East Lansing, MI, U.S.A.

BILLY LEE TURNER II
Clark University, Worcester, MA, U.S.A.

and

MARK A. COCHRANE
Michigan State University, East Lansing, MI, U.S.A.

Springer
# TABLE OF CONTENTS

Editors: Abbreviated Profiles .................................................. viii

List of Contributors .............................................................. xi

Foreword by Garik Gutman ..................................................... xxi

Section I  LCLUC Concepts; National and International Programs

1. The Development of the International Land Use and Land Cover Change (LUCC) Research Program and Its Links to NASA’s Land Cover and Land Use Change (LCLUC) Initiative
   Emilio F. Moran, David L. Skole, and B.L. Turner II .......................... 3

2. The NASA Land Cover and Land Use Change Program
   Garik Gutman, Christopher Justice, Ed Sheffner, and Tom Loveland ........ 19

3. Meeting the Goals of GOFC: an Evaluation of Progress and Steps for the Future
   John R. Townshend, Christopher O. Justice, David L. Skole, Alan Belward, Anthony Janetos, Iwan Gunawan, Johan Goldammer, and Bryan Lee ........... 33

Section II  Observations of LCLUC: Case Studies

   Introduction – Observations of LCLUC in Regional Case Studies
   David L. Skole and Mark A. Cochrane ........................................ 57

4. Forest Change and Human Driving Forces in Central America
   Steven A. Sader, Rinku Roy Chowdhury, Laura C. Schneider, and B. L. Turner II ................................................................. 61

5. Pattern to Process in the Amazon Region: Measuring Forest Conversion, Regeneration and Degradation
   David L. Skole, Mark A. Cochrane, Eraldo A. T. Matricardi, Walter Chomentowski, Marcos Pedlowski, and Danielle Kimble ............................... 81

6. Towards an Operational Forest Monitoring System for Central Africa
   Nadine T. Laporte, Tiffany S. Lin, Jacqueline Lemoigne, Didier Devers, and Miroslav Honzák ................................................................. 101

7. Land Use and Land Cover Change in Southeast Asia
   Jay H. Samek, Do Xuan Lan, Chaowalit Silapathong, Charlie Navanagruha, Sharifah Masturah Syed Abdullah, Iwan Gunawan, Bobby Crisostomo, Flaviana Hilario, Hoang Minh Hien, David L. Skole, Walter Chomentowski, William A. Salas, and Hartanto Sanjaya ........................................... 115
   Olga N. Krankina, Guoqing Sun, Herman H. Shugart, Vyacheslav Kharuk,
   Eric Kasischke, Kathleen M. Bergen, Jeffrey G. Masek, Warren B. Cohen,
   Doug R. Oetter, and Maureen V. Duane ................................. 127

9. Land Cover Disturbances and Feedbacks to the Climate System in
   Canada and Alaska
   A.D. McGuire, M. Apps, F.S. Chapin III, R. Dargaville, M.D. Flannigan,
   E. S. Kasischke, D. Kicklighter, J. Kimball, W. Kurz, D.J. McRae,
   K. McDonald, J. Melillo, R. Myneni, B.J. Stocks, D. L. Verbyla,
   and Q. Zhuang .............................................................. 143

10. Mapping Desertification in Southern Africa
    Stephen D. Prince ..................................................... 167

11. Woodland Expansion in U.S. Grasslands: Assessing Land-Cover
    Change and Biogeochemical Impacts
    Carol A. Wessman, Steven Archer, Loretta C. Johnson, and Gregory
    P. Asner ................................................................. 189

12. Arid Land Agriculture in Northeastern Syria: Will this be a tragedy of
    the commons?
    Frank Hole and Ronald Smith ......................................... 213

13. Changes in Land Cover and Land Use in the Pearl River Delta, China
    Karen C. Seto, Curtis E. Woodcock, and Robert K. Kaufmann ........ 227

Section III Cross Cutting Themes, Impacts and Consequences

14. The Effects of Land Use and Management on the Global Carbon Cycle
    R.A. Houghton, Fortunat Joos, and Gregory P. Asner .................. 243

15. Land Use and Hydrology
    John F. Mustard and Thomas R. Fisher ................................ 263

16. Land Use Change and Biodiversity: A Synthesis of Rates and
    Consequences during the Period of Satellite Imagery
    Andrew J. Hansen, Ruth S. DeFries, and Woody Turner .............. 283

17. Land Use and Climate
    Gordon B. Bonan, Ruth S. DeFries, Michael T. Coe, and Dennis S. Ojima . . . 307
18. **Urbanization**  
*Christopher D. Elvidge, Paul C. Sutton, Thomas W. Wagner, Rhonda Ryzner, James E. Vogelmann, Scott J. Goetz, Andrew J. Smith, Claire Jantz, Karen C. Seto, Marc L. Imhoff, Y. Q. Wang, Cristina Milesi and Ramakrishna Nemani.*  

19. **Land Use and Fires**  

20. **Land Cover / Use and Population**  
*Ronald R. Rindfuss, B. L. Turner II, Barbara Entwisle, and Stephen J. Walsh.*  

**Section IV**  
**Methodological Issues, Modeling**  

21. **Trends in Land Cover Mapping and Monitoring**  
*Curtis E. Woodcock and Mutlu Ozdogan.*  

22. **Linking Pixels and People**  
*Ronald R. Rindfuss, Stephen J. Walsh, B. L. Turner II, Emilio F. Moran, and Barbara Entwisle.*  

23. **Modeling Land-Use and Land-Cover Change**  
*Daniel G. Brown, Robert Walker, Steven Manson, and Karen Seto.*  

**Section V**  
**Synthesis and Lessons: Biophysical Change and Beyond**  

24. **Land-Use and Land-Cover Change Pathways and Impacts**  
*John F. Mustard, Ruth S. DeFries, Tom Fisher, and Emilio Moran.*  

25. **Integrated Land-Change Science and Its Relevance to the Human Sciences**  
*B. L. Turner II, Emilio Moran, and Ronald Rindfuss.*  

26. **Research Directions in Land-Cover and Land-Use Change**  
*Anthony C. Janetos.*
EDITORS:
Abbreviated Profiles

**Garik Gutman** is the Manager of the NASA Land-Cover/Land-Use Change Program at NASA’s Earth Science Enterprise. Prior to his NASA managerial position, his 14-years of research at NOAA included developing advanced algorithms for deriving land surface characteristics. His research interests include the use of remote sensing for detecting changes in land cover and land use, and analyzing the impacts of these changes on climate and environment. His NASA research program is helping develop the underpinning science for land cover and land use change and promotes scientific international cooperation through supporting the development of regional science networks over the globe.

**Anthony Janetos** is the Vice President of the H. John Heinz III Center for Science, Economics and the Environment. Previously, he served as Vice President for Science and Research at the World Resources Institute, and Senior Scientist for the Land-Cover and Land-Use Change Program in NASA’s Office of Earth Science. He also was Program Scientist for NASA’s Landsat 7 mission. Dr. Janetos has many years of experience in managing scientific and policy research programs on a variety of ecological and environmental topics, including air pollution effects on forests, climate change impacts, land-use change, ecosystem modeling, and the global carbon cycle. He was a co-chair of the U.S. National Assessment of the Potential Consequences of Climate Variability and Change and an author of the IPCC Special Report on Land-Use Change and Forestry and the Global Biodiversity Assessment. Most recently he has served on National Research Council Committees on Funding Scientific Research at the Smithsonian Institution and Reviewing the Bush Administration’s Climate Change Science Strategic Plan.

**Christopher O. Justice** received his PhD from the University of Reading, UK. He undertook Postdoctoral Fellowships with the NRC at NASA Goddard Spaceflight Center and with the European Space Agency at ESRIN, Frascati. He joined the University of Maryland and became a visiting scientist at the Goddard Spaceflight Center where he participated in the early development of moderate resolution remote sensing with the GIMMS group. He took a position as Research Professor in the Department of Environmental Sciences at the University of Virginia and in 2001 accepted a position in the Geography Department, University of Maryland as Professor and Research Director. He is the Program Scientist for the NASA’s Land Cover and Land Use Change Program, a team member and land discipline leader of the NASA Moderate Imaging Spectroradiometer (MODIS) Science Team and is responsible for the MODIS Fire Product and the MODIS Rapid Response System. He is a member of the NASA NPOESS Preparatory Project (NPP) Science Team. He has been instrumental in the development the new Land Use Land Cover Change (LULCC) element of the USGCRP and is co-chair of the GOFC/GOLD-Fire Implementation Team, a project of the Global Terrestrial Observing System (GTOS).
Emilio F. Moran is the Rudy Professor of Anthropology, Professor of Environmental Sciences, and Adjunct Professor of Geography at Indiana University, Bloomington, IN USA. Dr. Moran is the Director of Anthropological Center for Training and Research on Global Environmental Change, www.indiana.edu/~act, Co-Director of Center for the Study of Institutions, Population and Environmental Change, www.cipec.org. He is also the Leader, Focus 1, LUCC. Land Use and Land Cover Change Programme is a joint core project of IGBP (The International Geosphere-Biosphere Programme) and IHDP (The International Human Dimensions Programme on Global Environmental Change), www.indiana.edu/~act/focus1. He is Co-Chair for THE LAND Project, IGBP phase II, and Regional Editor for the Americas of the journal Land Degradation and Development.

John F. Mustard is an Associate Professor in the Department of Geological Sciences and the Center for Environmental Studies at Brown University, USA. His research interests include land surface dynamics and the relationship to physical and anthropogenic drivers, with an emphasis on ecosystem interactions to change in water quality and quantity in arid and semi-arid regions.

Ronald R. Rindfuss is a Professor in the Department of Sociology and a Fellow at the Carolina Population Center, University of North Carolina, Chapel Hill. His research interests include fertility, aspects of the life course and population and the environment. He is currently working on projects in Thailand, Norway, Japan and the United States.

David Skole is currently the director of the Basic Science and Remote Sensing Initiative, a research program focused on environmental research using remote sensing systems. Dr. Skole's research interests focus on the role humans play in changing land cover throughout the world. Currently he is involved in research projects focused on understanding the interannual variation in deforestation rates, and the social and ecological controls on its variation over time. He is Principal Investigator of a NASA ESIP center. He is PI for the NASA Landsat Pathfinder Project and the PI on a number of other funded research projects including the Large Scale Amazon Basin Experiment. He is a member of the Landsat 7 Science Team, and is a PI with the Canadian Radarsat program and the Japanese JERS program. Dr. Skole is the PI on the NASA funded Center of Excellence in the Applications of Remote Sensing at Michigan State University. Dr. Skole is the Chairman of the IGBP/IHDP Core Project on Land Use and Cover Change, a Steering Committee member for the IGBP Data and Information Systems project, as well as a member of the Standing Committees of the IGBP and IHDP. He has served on several NASA committees and panels for EOS and its data system and other programs. He is currently the High Resolution Design Team Leader for the CEOS project on Global Observations of Forest Cover.

B. L. Turner II is the Higgins Professor of Environment and Society, Graduate School of Geography and George Perkins Marsh Institute, Clark University, Worcester, MA USA. His research interests are human-environment relationships, especially land-change science. He is currently working on projects in Yucatan, Mexico, and central Massachusetts.
Mark A. Cochrane received his PhD Ecology from The Pennsylvania State University, and his S.B. Environmental Engineering Science from the Massachusetts Institute of Technology. He is currently a researcher and supervisor for projects using remote sensing to investigate land-use and land-cover change in tropical and temperate regions. Additionally he is working on the ecological effect of disturbance (e.g. logging and fire) on forest ecosystems. His publications encompass papers on fire and deforestation in Amazonian forests, and the future of the Brazilian Amazon.
LIST OF CONTRIBUTORS

M. APPS
Canadian Forest Service, Pacific Forestry Centre, 506 West Burnside Road, Victoria, British Columbia, V8Z 1M5 Canada

STEVEN ARCHER
University of Arizona, School of Renewable Natural Resources, 325 Biological Sciences East Bldg., P.O. Box 210043, Tucson, Arizona 85721-0043, USA

GREGORY P. ASNER
Carnegie Institution of Washington, Department of Global Ecology, 260 Panama Street, Stanford, California 94305, USA
gasner@globalecology.stanford.edu

ALAN BELWARD
Space Applications Institute, Global Vegetation Monitoring Unit, Joint Research Center-TP261, I-21020, Ispra, Italy
alan.belward@jrc.it

KATHLEEN M. BERGEN
The University of Michigan, School of Natural Resources and Environment, 430 E. University, Ann Arbor, MI 48109-1115, USA

GORDON B. BONAN
National Center for Atmospheric Research , P.O. Box 3000, Boulder, CO 80307, USA

DANIEL G. BROWN
University of Michigan, School of Natural Resources and Environment, Ann Arbor, MI 48109, USA, danbrown@umich.edu

F. BROWN
Universidade Federal do Acre Rio Branco, UFAC/WHRC Setor do Estudos do Uso da Terra e Mudanças Globais – SETEM Parque Zoobotanico Acre 69.915-900, Brazil

F. STUART CHAPIN III
University of Alaska Fairbanks, Department of Biology and Wildlife, Institute of Arctic Biology, Fairbanks, Alaska, 99775, USA
terry.chapin@uaf.edu

WALTER CHOMENTOWSKI
Michigan State University, Center for Global Change and Earth Observations, East Lansing, MI 4882, USA

RINKU ROY CHOWDHURY
Clark University, Graduate School of Geography and Marsh Institute, Worchester, MA 01610, USA
MARK A. COCHRANE
Michigan State University, Center for Global Change and Earth Observations, 1405 S. Harrison Road, Room 101, East Lansing, MI 48823-5243, USA

MICHAEL T. COE
University of Wisconsin-Madison, Center for Sustainability and the Global Environment, Gaylord Nelson Institute for Environmental Studies, 1710 University Ave., Madison, WI 53726, USA

WARREN B. COHEN
USDA Forest Service, PNW Research Station, 3200 SW Jefferson Way, Corvallis, OR 97331, USA

S.G. CONARD
USDA Forest Service, 1601 North Kent Street, Arlington, VA 22209 USA

BOBBY CRISOSTOMO
Database Management Division, National Mapping and Resource Information Authority, Manila, Philippines

I. CSISZAR
University of Maryland, Department of Geography, 2181 LeFrak Hall, College Park, MD 20742, USA

R. DARGAVILLE
Laboratoire des Sciences du Climat et de l’Environnement (LSCE) CEA/CNRS, Paris, France

RUTH S. DEFINES
University of Maryland, Department of Geography and Earth Systems Science Interdisciplinary Center, Computer and Space Sciences Building and Department of Geography, 2181 Lefrak Hall, College Park, MD 20742, USA

DIDIER DEVER
University of Maryland, Department of Geography, 2181 LeFrak Hall, College Park, MD 20742 USA

MAUREEN V. DUANE
Oregon State University, Department of Forest Science, 202 Richardson Hall, Corvallis, OR 97331-5752, USA

CHRISTOPHER D. ELVIDGE
NOAA National Geophysical Data Center, 325 Broadway, Boulder, Colorado 80303, USA. chris.elvidge@noaa.gov

BARRABARA ENTWISLE
University of North Carolina at Chapel Hill, Chapel Hill, NC 27599, USA
THOMAS R. FISHER
University of Maryland, Horn Point Laboratory, Center for Environmental Science,
Cambridge, MD 21613, USA

M.D. FLANNIGAN
Canadian Forest Service, Great Lakes Forestry Centre, 1219 Queen Street East, Sault
Ste. Marie, Ontario, P6A 2E5 Canada

P.G.H. Frost
University of Zimbabwe, Institute of Environmental Studies, P.O. Box MP 167, Mount
Pleasant, Harare, Zimbabwe

L. Giglio
Science Systems and Applications, Inc., NASA/GSFC, Code 923 Greenbelt, MD 20771

SCOTT J. GOETZ
University of Maryland, Department of Geography, 2181 LeFrak Hall, College Park,
MD 20742, USA

JOHANN GEORG GOLDMAMMER
Global Fire Monitoring Center (GFMC), Max Planck Institute for Chemistry,
Biochemistry Department, c/o University of Freiburg, George Koehler Allee 75, 79110
Freiberg, Germany
johann.goldammer@fire.uni-freiburg.de

IWAN GUNAWAN
Agency for the Assessment and Application of Technology (BPPT), Bogor, Indonesia
and
Bureau of Programme Coordination and External Relations, ASEAN Secretariat,
Jakarta Indonesia

GARIK GUTMAN
NASA Headquarters, 300 E Street, SW, Washington, DC 20546, USA
ggutman@hq.nasa.gov

ANDREW J. HANSEN
Montana State University, Ecology Department, Bozeman, MT 59717, USA

HOANG MINH HIEN
Disaster Management Center, Standing Office of the Central Committee for Flood and
Storm Control, Hanoi, Vietnam

FLAVIANA HILARIO
Climatology and Agrometeorology Branch, Philippine Atmospheric, Geophysical and
Astronomical Services Administration, Quezon City, Philippines

FRANK HOLE
Yale University, Anthropology, New Haven, CT 06520, USA
MIROSLAV HONZÁK
Conservation International, 1919 M Street, NW, Washington, DC 20036 USA

R.A. HOUGHTON
Woods Hole Research Center, Woods Hole, MA 02543, USA
rhoughton@whrc.org

MARC L. IMHOFF
NASA Goddard Space Flight Center, Greenbelt, Maryland USA

ANTHONY C. JANETOS
The H. John Heinz III Center for Science, Economics and the Environment, 1001 Pennsylvania Ave., NW, Suite 735 South Washington, DC 20004, USA
janetos@heinzctr.org

CLAIREF JANTZ
University of Maryland, Department of Geography, 2181 LeFrak Hall, College Park, MD 20742, USA

LORETTA C. JOHNSON
Kansas State University, Division of Biology, Ackert Hall Rm 232, Manhattan, Kansas 66506, USA

FORTUNAT JOOS
University of Bern, Climate and Environmental Physics, Physics Institute, Sidlerstr. 5, CH-3012 Bern, Switzerland
joos@climate.unibe.ch

CHRISTOPHER O. JUSTICE
University of Maryland, Department of Geography, 2181 LeFrak Hall, College Park, MD 20742 USA

ERIC S. KASISCHKE
University of Maryland, Department of Geography, 2181 LeFrak Hall, College Park, MD 20742, USA

ROBERT K. KAUFMANN
Boston University, Department of Geography & Center for Environmental Science and Policy, Boston, MA, USA

VYACHESLAV KHARUK
Forest Biophysics Lab, V. N. Sukachev Institute of Forest, Academgorodok, Krasnoyarsk, 660036 Russia

D. KICKLIGHTER
The Ecosystems Center, Marine Biological Lab, Woods Hole, MA, USA
J. KIMBALL
University of Montana, Flathead Lake Biological Station, Missoula, MT, USA

DANIELLE KIMBLE
Michigan State University, Center for Global Change and Earth Observations, East Lansing, MI 4882, USA

OLGA N. KRANKINA
Oregon State University, Department of Forest Science, 202 Richardson Hall, Corvallis, OR 97331-5752, USA

W. KURZ
Canadian Forest Service, Pacific Forestry Centre, Victoria, British Columbia, Canada

DO XUAN LAN
Forest Inventory and Planning Institute, Ministry of Agricultural and Rural Development, Hanoi, Vietnam

NADINE T. LAPORTE
The Woods Hole Research Center, P.O. Box 296, Woods Hole, MA 02543 USA

BRYAN LEE
Canadian Forest Service, Edmonton, Canada

JACQUELINE LEMOIGNE
NASA Goddard Space Flight Center, Applied Information Sciences Branch, Code 935, Building 28, Greenbelt, MD 20771 USA

TIFFANY S. LIN
The Woods Hole Research Center, P.O. Box 296, Woods Hole, MA 02543 USA

TOM LOVELAND
USGS Eros Data Center, Sioux Falls, SD, USA

STEVEN MANSON
University of Minnesota, Department of Geography, Minneapolis, MN 55455, USA

JEFFREY G. MASEK
NASA GSFC, Code 923, Greenbelt, MD 20771, USA

SHARIFAH MASTURAH SYED ABDULLAH
Earth Observation Centre, Universiti Kebansaen Malaysia, Selangor, Malaysia

ERALDO A. T. MATRICARDI
Michigan State University, Center for Global Change and Earth Observations, East Lansing, MI 4882, USA
D.J. MCRAE
Canadian Forest Service, Great Lakes Forestry Centre, 1219 Queen Street East, Sault Ste. Marie, Ontario, P6A 2E5 Canada

K. MCDONALD
NASA Jet Propulsion Laboratory, Pasadena, CA, USA

A.D. McGuire
University of Alaska Fairbanks, U.S. Geological Survey, Alaska Cooperative Fish and Wildlife Research Unit, 214 Irving I, Fairbanks, AK 99775 USA

J. MELILLO
The Ecosystems Center, Marine Biological Lab, Woods Hole, MA, USA

CRISTINA MILESI
University of Montana, Missoula, Montana, USA

EMILIO F. MORAN
Indiana University, ACT, Anthropological Center for Training and Research on Global Environmental Change, 701 E. Kirkwood Ave., Student Building 331, Bloomington, IN 47405-7710, USA
moran@indiana.edu

JOHN F. MUSTARD
Brown University, Department of Geological Sciences, Box 1846, Providence, RI 02912, USA

R. MYNENI
Boston University, Department of Geography, Boston, MA, USA

CHARLIE NAVANAGRUAH
Mahidol University, Bangkok, Thailand

RAMAKRISHNA NEMANI
University of Montana, Missoula, Montana, USA

DOUG R. OETTER
Georgia College & State University, Department of History and Geography, Milledgeville, GA 31061-0490, USA

DENNIS S. OJIMA
Colorado State University, Natural Resource Ecology Laboratory, NESB, B229, Fort Collins, CO 80523, USA

MUTLU OZDOGAN
Boston University, Department of Geography, 675 Commonwealth Ave., Boston, MA 02215, USA
MARCOS PEDLOWSKI
Universidade Estadual do Norte Fluminense, Rio de Janeiro, Brazil

STEPHEN D. PRINCE
University of Maryland, Geography Department, Room 2181, LeFrak Hall, College Park, MD 20742-8225, USA

RONALD R. RINDFUSS
University of North Carolina at Chapel Hill, Department of Sociology, Campus Box 3210, Hamilton Hall, Chapel Hill, NC 27599-3210, USA
ron_rindfuss@unc.edu

D.P. ROY
University of Maryland, Department of Geography, College Park, and NASA Goddard Space Flight Center, Code 922, Greenbelt, MD 20771 USA

T.S. RUPP
University of Alaska Fairbanks, School of Agriculture and Land Resource Management, P.O. Box 757140 Fairbanks, AK 99775, USA

RHONDA RYZNER
Tufts University, Boston, MA, USA

STEVEN A. SADER
University of Maine, Department of Forest Management, Orono, ME 04469-5755, USA

WILLIAM A. SALAS
Applied Geosolutions, LLC, Durham, NH, USA

JAY H. SAMEK
Michigan State University, Center for Global Change and Earth Observations, East Lansing, MI 4882, USA

HARTANTO SANJAYA
Agency for Assessment and Application of Technology (BPPT), Jakarta, Indonesia

LAURA C. SCHNEIDER
Clark University, Graduate School of Geography and Marsh Institute, Worcester, MA 01610, USA

KAREN C. SETO
Stanford University, Center for Environmental Science and Policy, Institute for International Studies, Stanford, CA 94305-6055, USA

ED SHEFFNER
NASA Headquarters, 300 E Street, SW, Washington, DC 20546, USA
HERMAN H. SHUGART  
University of Virginia, Department of Environmental Sciences, Box 400123, 
Charlottesville, Virginia 22904, USA

CHAOWALIT SILAPATHONG  
Geo-Informatics And Space Technology Development Agency, Bangkok, Thailand 

DAVID L. SKOLE  
Michigan State University, Department of Geography, Center for Global Change and 
Earth Observations, 218 Manly Miles Building, East Lansing, MI 48823, USA

ANDREW J. SMITH  
University of Maryland, Department of Geography, 2181 LeFrak Hall, College Park, 
MD 20742, USA

RONALD SMITH  
Yale University, Geology & Geophysics, New Haven, CT 06520, USA

B.J. STOCKS  
Canadian Forest Service, Great Lakes Forestry Centre, 1219 Queen Street East, Sault 
Ste. Marie, Ontario, P6A 2E5 Canada

GUOQING SUN  
University of Maryland, NASA GSFC, Biospheric Sciences Branch 
Greenbelt, MD 20771, USA

PAUL C. SUTTON  
University of Denver, Department of Geography, Denver, Colorado USA

JOHN R. TOWNSHEND  
University of Maryland, Department of Geography, 2181 LeFrak Hall, College Park, 
MD 20742 USA

B. L. TURNER II  
Clark University, Graduate School of Geography & George Perkins Marsh Institute, 
Worcester, MA 01610, USA  
bturner@clarku.edu

WOODY TURNER  
NASA Office of Earth Science, Mail Code YS, Washington, DC 20546, USA

D.L. VERBYLA  
University of Alaska Fairbanks, School of Agriculture and Land Resource 
Management, P.O. Box 757140 Fairbanks, AK 99775, USA

JAMES E. VOGELMANN  
SAIC, USGS Eros Data Center, Sioux Falls, South Dakota, USA
THOMAS W. WAGNER
University of Michigan, China Data Center, Ann Arbor, Michigan USA

ROBERT WALKER
Michigan State University, Department of Geography, East Lansing, MI 48824, USA

STEPHEN J. WALSH
University of North Carolina at Chapel Hill, Chapel Hill, NC 27599, USA

Y.Q. WANG
University of Rhode Island, Department of Natural Resources, Kingston, RI 02881, USA

CAROL A. WESSMAN
University of Colorado, Cooperative Institute for Research in Environmental Sciences & Department of Ecology and Evolutionary Biology, CIRES 216 UCB, Boulder, Colorado 80309-0216, USA

CURTIS E. WOODCOCK
Boston University, Department of Geography & Center for Remote Sensing, 675 Commonwealth Ave., Boston, MA 02215, USA

Q. ZHUANG
The Ecosystems Center, Marine Biological Lab, Woods Hole, MA, USA
FOREWORD

Land-Use and Land-Cover Change (LULCC) research has become an important element of global change research programs at national as well as international levels. Recently, LULCC was established as a separate element of the United States Climate Change Science Program and is currently being further developed by a US interagency effort.

During the past 8 years NASA has supported several dozen projects devoted to the topic of land-cover and land-use change. The NASA Land-Cover/Land-Use Change (LCLUC) Program started as a combination of regional satellite-based studies, representative field-based process studies and modeling. It has grown into a large, multifaceted, interdisciplinary science program, linked to other NASA programs, such as Terrestrial Ecology, Terrestrial Hydrology, Water Cycle, Carbon Cycle, Ocean Biology, and Applications. on projects that comprised the first phase of the program (1996-2001). It represents a compilation of state-of-the-art research results devoted to changes in land cover.

This book is a collection of papers based on the projects conducted under the NASA LCLUC program during recent years with an emphasis and land use, an analysis of their causes, methods for their detection, their monitoring, their impacts on climate and environment and lessons learned. The volume comprises five Sections. Section I describes major LULCC concepts and recent developments under the NASA LCLUC Program and other programs at both national and international levels. Results of the case studies in various geographic regions are examined in Section II. Section III is devoted to cross-cutting themes and consequences of changes in land cover and land use on carbon and water cycles, biodiversity, climate, urbanization, fire, population, and land resources. Methodological issues and LULCC modeling are described in Section IV. Finally, Section V is an attempt to synthesize the accumulated knowledge of the lessons learned.

LULCC science is truly interdisciplinary. It requires an alliance of physical scientists (e.g. geographers, climatologists, ecologists, hydrologists), remote sensing scientists) and social scientists (e.g. economists, demographers, human geographers, anthropologists). Because LULCC breaks new ground in interdisciplinary research and the emergence of a new research paradigm -- Integrated Land-Change Science -- it has made important contributions to methods on land-cover change detection and classification, on linking people and pixels, on the synergistic use of optical and radar satellite data, and in modeling the dynamic process of land-use and land-cover change. This volume is prepared by scientists from different backgrounds that have teamed up in a joint effort to write the chapters and bring into one manuscript the major recent advances in LCLUC science. The book will be of interest to those involved in studying changes in land cover, be it senior scientists, young scientists or students.

Garik Gutman
Section I  LCLUC Concepts; National and International Programs
CHAPTER 1

THE DEVELOPMENT OF THE INTERNATIONAL LAND-USE AND LAND-COVER CHANGE (LUCC) RESEARCH PROGRAM AND ITS LINKS TO NASA’S LAND-COVER AND LAND-USE CHANGE (LCLUC) INITIATIVE

EMILIO F. MORAN\(^1\), DAVID L. SKOLE\(^2\), B. L. TURNER II\(^3\)

\(^1\)Indiana University, Anthropological Center for Training and Research on Global Environmental Change, Bloomington, IN 47405
\(^2\)Michigan State University, Center for Global Change and Earth Observations, East Lansing, MI 48823
\(^3\)Clark University, Graduate School of Geography & George Perkins Marsh Institute, Worcester, MA 01610

1 Introduction

The study of land-use and land-cover change has a long history dating to ancient times. (Glacken 1967)\(^1\) Early concern focused on how human activities transformed and degraded landscapes, a theme that has resurfaced at various times (Marsh 1864; Thomas 1956; Moran 2000) and currently is embedded within the larger concept of global environmental change and earth system science, especially that part addressing land-use and land-cover change (Meyer and Turner, 1994; Turner, Steffen et al., 2002).

It is unquestionable that human populations have affected the structure and function of the earth system since their evolution as a distinct modern species (Thomas 1956; Redman 1999), but this impact increased in pace, magnitude, and kind with the advent of the industrial revolution (Turner et al., 1990; Meyer 1996; Steffen et al., 2003). Human-induced changes in the terrestrial surface of the earth have been substantial, especially deforestation (Watson et al., 2001; Williams 2003), and they have affected the delivery of ecosystem services and contributed to altered biogeochemical cycles that control the functioning of the earth system (Steffen et al. 2003). The human imprint on many “natural” conditions and processes is so large that separating the natural from the human not only proves difficult but analytically questionable, especially in regard to terrestrial processes (Vitousek et al., 1997; Clark et al., 2003).

To be sure, land-cover and land-use change is only one component of global environmental changes currently underway, and is superceded by fossil fuel consumption in regard to atmospheric warming (Steffen et al., 2001). Energy use, however, is tightly linked to population and its standards of consumption, and this linkage interacts with socio-political and cultural structures to create pressure on land users to produce more goods and services to meet human demands. The sources of this demand and the location of production to meet it are not necessarily spatially congruent, and large regional differences in access to land and land-based resources exist. It is precisely these kinds of disconnects and discrepancies in land change and its

\(^1\) The literatures documenting our observations of the history of human-environment relationships is very large and its full account beyond the confines of this volume. The few references selected, therefore, serve as guiding examples and are by no means exhaustive.
various consequences that require an understanding of land-use and land-cover change in which its global and local-regional dimensions are connected.

This understanding requires linkages between the biophysical and human dimensions of land cover and land use. Land cover refers to the land’s physical attributes (e.g., forest, grassland), whereas land use expresses the purpose to which those attributes are put or how they are transformed by human action (e.g., cropping, ranching). As this volume demonstrates, land cover is visible in remotely-sensed data from satellite platforms, although it requires interpretation and ground-truthing. In general, use of satellite imagery for fine-resolution analysis increases the need for detailed ground-based land-use information. Regardless, land cover and land use are so intimately linked that understanding of either requires a coupled human-environment system analysis. After all, the entire terrestrial surface of the earth is claimed by someone, and significant portions of it are actively managed.

2 The Development of Contemporary Land-Use and Land-Cover Change Science

The science of global environmental change has, arguably, been responsible for the discovery of the rapid and large-scale accumulation of CO$_2$ in the atmosphere and the concern that this process will trigger global climate changes whose consequences could threaten the planet. Research quickly identified land-use and land-cover changes as a major element of the global carbon cycle, both as source and sink (Moore et al., 1981; Houghton et al., 1983; Woodwell et al., 1983). This role in the carbon cycle turned research interests in land change towards the human alteration and conversion of landscapes, especially forests, agricultural lands, and grasslands, which increased or reduced carbon in the atmosphere. In addition, attempts to balance the carbon cycle identified land cover as a candidate for helping explain the so-called missing carbon sink, with recent evidence pointing to such land changes as the regeneration of forests on abandoned agricultural lands as well as changes in ecosystem production due to longer growing seasons and fertilization by CO$_2$ and nitrogen (Schimel et al., 2001; Goodale et al., 2002). With these questions as points of departure, the reach of global environmental change research subsequently expanded to include a broad array of human-induced changes in structure and function of the earth system, including ecosystems and their services and biodiversity (Lubchenco 1998; Daily et al., 2000; Raven 2002) in which land change plays a critical, if not fundamental role. Recent evidence points to the importance of regional-to-local climate change as driven by land change (Kalnay and Cai, 2003), and the emergence of sustainability science (Kates et al., 2001; Clark et al., 2003) adds yet another strong interest in land change, with strong policy implications (Turner et al., 2003).

At the same time, there were significant advances in the use of earth observation data and information to support the science of global change and sustainability. Global scale datasets from coarse resolution sensors were making it possible to monitor and measure changes in land cover, including phenology, net primary production (NPP), and other dynamic properties (Justice et al., 1985; Tucker et al., 1985; Sellers et al., 1994; Townshend et al., 1994). Similar advances were made in the use of fine resolution earth observation data for quantification of land cover conversion rates, and to a more limited extent to assess dynamics of land use change.
DEVELOPMENT OF LUCC AND ITS LINK TO LCLUC

over continental sized areas and at watershed scales (Skole and Tucker, 1993; Skole et al., 1994; Brondizio et al., 1996; Batistella et al., 2003).

Thus, land-use and land-cover change has emerged as one of the key independent themes in the global change, climate change, earth systems, and sustainability research programs. Advances in large-area measurements from remote sensing, increased sophistication of process-level analyses from case studies, and in modeling are evidence of significantly improved capability within the research community. Land change is now recognized as a topic of study in its own right, requiring a concerted and focused program to document and understand its causes and consequences. This intellectual history of the science informs the programmatic or institutional history of land-use and land-cover change.

The institutional history of research programs devoted to land change begins with the recognition by natural and remote sensing scientists engaged with the International Geosphere-Biosphere Programme (IGBP) that understanding land-cover dynamics, be they ecosystem or climate change, was difficult in the absence of a complementary understanding of land-use dynamics. The latter, in turn, required social science expertise as it involved assessing how people made decisions about land. A joint effort among the natural, social and remote sensing sciences seemed the best means of achieving integrated understanding of land change. With this in mind, the IGBP approached the International Social Science Council (ISSC) to put together a “working group” (B. L. Turner II, chair, and David Skole, co-chair) to explore the possibility of creating a joint core project/research program at the international level to be shared between the two entities.

This effort took place in concert with various national and international efforts to broaden global environmental change research beyond climate change per se and to develop a research agenda on the human dimensions of this change. In the U.S., an effort to stake out this agenda was begun in 1989 by a committee of the Social Science Research Council (SSRC) whose deliberations and reports informed the U.S. National Research Council (NRC) and its then newly established Committee on the Human Dimensions of Global Change (HDGC), chaired by Oran Young. At the same time, Harold Jacobson led an effort sponsored by the ISSC that resulted in the creation of an international Human Dimensions Program (HDP), based originally in Geneva, Switzerland, and which later became the IHDP, currently based in Bonn, Germany. Each of these committees staked out a full range of human dimensions issues and potential research programs, exemplified in the NRC’s committee volume, Global Change: Understanding the Human Dimensions (Stern et al., 1991). Importantly, each of these committees identified land-use and land-cover change as the top priority research topic wherein a joint effort between the natural, social, and remote sensing-geographical information sciences was most likely to pay off in the immediate future.

With this backing the IGBP-ISSC’s working group recommended the creation of a core project on LUCC (Turner et al., 1993) and identified the broad course of research that it might pursue. A Core Project Planning Committee (CPPC-IGBP)/Research Project Planning Committee (RPPC-HDP) was established to create a LUCC Science Plan to guide the work, retaining Turner and Skole in chair and co-chair capacities. During this process the international human dimensions program became jointly sponsored by ISSC and the International Council of Scientific Unions (ICSU, subsequently renamed the International Council for Science but retaining the acronym,
ICSU), the latter organization the sponsor of the IGBP. This tie facilitated a union of the IGBP and IHDP in supporting LUCC.

To produce the science plan, the CPPC/RPPC held meetings worldwide with different communities of researchers as well as maintaining linkages in the US with the NRC. The LUCC Science Plan (Turner et al., 1995) defined several major science questions which have been central to the joint core project during its tenure:

- How has land cover changed over the last 300 years as a result of human activities?
- What are the major human causes of land cover change in different geographical and historical contexts?
- How will changes in land use affect land cover in the next 50 to 100 years?
- How do immediate human and biophysical dynamics affect the sustainability of specific types of land uses?
- How might changes in climate and global biogeochemistry affect both land use and land cover?
- How do land uses and land covers affect the vulnerability of land users in the face of change and how do land cover changes in turn impinge upon and enhance vulnerable and at-risk regions?

The LUCC project was formally inaugurated by a 1996 Open Science Meeting in Amsterdam, hosted by the Royal Netherlands Academy of Sciences (Fresco et al., 1997). The first LUCC scientific committee was chaired by David Skole and the International Project Office (IPO) was based at the Institut Cartogràfic de Catalunya in Barcelona. During this period LUCC joined the IGBP’s Global Change and the Terrestrial Ecosystem (GCTE) project in hosting a joint GCTE-LUCC international science conference in Barcelona in 1998. This well-attended meeting demonstrated the strong potential of natural science, social science, and remote sensing/GIS science communities to create an integrated science of land change and began a process of setting initial priorities for implementation of LUCC. Subsequently, Eric Lambin became the second chair of LUCC, overseeing the formulation of a LUCC Implementation Plan (Lambin et al. 1999) and an enlargement of the project’s research objectives outlined in the earlier Science Plan. In 2000 the IPO moved to Belgium with support from the Belgian Government, the University of Louvain-le-Neuve, and IHDP, where it currently resides. The IPO and its three research foci offices (land use dynamics, land cover dynamics, integrative regional and global modeling) galvanize and network land-change research worldwide and undertake synthesis activities (e.g., Lambin et al., 2001; McConnell and Moran, 2001; Geist and Lambin, 2002; Parker et al., 2002).

These planning and agenda-setting activities and the early research projects that were initiated by the research community helped to foster land-change funding programs within agencies and organizations worldwide, in many cases cooperating with IAI (Inter-American Institute for Global Change Research), APN (Asian-Pacific Network for Global Change Research), START (Global Change System for Analysis, Research and Training), and GCTE (global change in terrestrial ecosystems project), among others. In the United States of America, for example, land-use and land-cover change was identified as one of the Grand Challenges in Environmental Sciences by the National Research Council (NRC 2001), and NASA developed its own Land-Cover
DEVELOPMENT OF LUCC AND ITS LINK TO LCLUC

and Land-Use Change research program (see details in Chapter 2, this volume), taking elements of its Science Plan from the international LUCC programs. The last venture involved the efforts of David Skole (then Chair of LUCC) and Chris Justice (then IGBP-DIS Focus 1 leader) working with NASA officers, and the first phase results of these efforts are addressed in this volume. Land-use and -cover change subsequently began to emerge in other programs, including NOAA and NSF, and with the writing of the science plan for the next decade of the US Global Change Research Program (USGCRP), land-change science was incorporated explicitly as one of the major themes (see below).

The LUCC core project will continue through 2005. Beginning in 2002, an effort began to develop a new generation land-centric project that would merge various other programs of the IGBP, especially those within GCTE, and parts of the IHDP into a new, integrated “Land Project” (see http://www.igbp.kva.se for details on the new IGBP phase II planning). This convergence of interests is motivated by the goal of developing truly integrative research on coupled human-natural systems and producing policy-relevant research that will enhance sustainability and reduce vulnerability of land systems.

3 Insights Gained: Examples

The land-change research community has made considerable progress during the formative stages of the formal international and national programs in question (Turner 2002). Indeed, the number and range of accomplishments are sufficiently large that we make no attempt to cover them in detail here and direct the reader to the in-depth treatment found in the various chapters on this volume which highlight NASA-LCLUC sponsored work in the context of the larger land-change community. Here we provide a brief overview made on three, not necessarily exclusive, research fronts: monitoring and observing land-cover change, land-use and land-cover dynamics, and land-change modeling.

3.1 MONITORING AND OBSERVING LAND CHANGE

Both the international programs and the agency programs in the US recognize the importance of observations. Early in the development of a research agenda, it was clear that data on rates of land-cover change were missing or inadequate to form a basis for more process-driven analysis. One of the critical challenges was obtaining global, consistent measurements of land cover and its change with known accuracy. A complementary challenge for land-use and -cover change analysis was in obtaining large-area observations, at the scale of regions or continents, at the spatial resolution needed to measure fine scale changes associated with such phenomena as deforestation, fire, and degradation. At the same time, significant advances were being made with NASA-supported projects in the area of global land-cover datasets and high resolution regional land-cover change datasets (DeFries et al., 1999; 2000). The IGBP-DIS developed DISCover, a global 1 km resolution land-cover dataset, using a system of coordinated ground station acquisitions and an international consensus effort to define processing and validation methods (Townshend et al., 1994). This precursor effort has led to further development of land cover monitoring using MODIS and other new
sensors (Justice and Townshend, 2002; Justice et al., 2002) with their implications for operational land cover monitoring (Townshend and Justice, 2002). The Landsat Pathfinder project focused on developing detailed measurements of rates of forest conversion in the tropics, North America, and in selected case study sites (Skole and Tucker, 1993; Steininger et al., 2001). These initial efforts focused on rather straightforward classification schemes, but provided important early direct estimates of important global change parameters.

Today, the land-use and land-cover change community has made significant advances in moving beyond classification to direct parameterization and measurement of continuous fields (Qi et al., 2000; Hansen et al., 2002). For example, work supported by the NASA LCLUC program produces fractional cover data and global percent tree cover datasets from AVHRR and MODIS, which have important significance for carbon cycle studies. Landsat is used in conjunction with fractional forest cover continuous fields analyses to measure and map forest degradation, hence moving the observational capabilities beyond the early forest-non-forest classification. In addition, important high temporal frequency land-cover changes are also monitored globally, such as fire—an important proximate cause of land-cover change. The community is also backfilling the historical record using tabular datasets from reference sources on distribution of cropland and other land covers (Ramankutty and Foley, 1999; Goldewijk 2001; Goldewijk and Ramankutty, 2003).

While global measurement and monitoring has been the intended goal, both the NASA programs and the research community have recognized the important role of regional networks and regional assessments. Through the development of networks of scientists in regions throughout the world, it is possible to improve both calibration and validation of products. Moreover, the regional context provides a framework for developing detailed case studies which provide an analytical approach to linking the patterns to the processes of change, with particular emphasis on drivers of land-use and -cover change. The importance of a regional framework is demonstrated in the initiation of several important science projects and campaigns in Amazonia, Southeast Asia, Central and South Africa and Northern Eurasia. These regional efforts have advanced the fundamental observations and science missions of the LCLUC programs and provided a framework for linking science to assessment, policy and capacity building.

3.2 LAND-USE AND LAND-COVER DYNAMICS

Observation-based efforts fuel the data used for a large array of individual research activities that have made considerable advances in the techniques and analysis used to address land change and its impact on global change. Coarse and moderate resolution data provide information for biophysical studies of net primary production and vegetation dynamics (Myneni et al., 2001; Tucker et al., 2001; Zhou et al., 2001; Nemani et al., 2003) and fine resolution data provide information to assess human impacts (DeFries et al., 2002; Taylor et al., 2002; Houghton et al., 2000). The science is now identifying important components of these dynamics, focusing on the interaction among multiple agents of land use/cover change. A recent example is the complex relationship between deforestation, selective logging and fire in the tropics (Cochrane et al., 1999; Nepstad et al., 1999b; Cochrane 2001). Land use and land cover change (in the tropics and elsewhere) arises by virtue of complex interactions, leads to unexpected
feedbacks, and broadcasts ecological impact beyond the boundaries of direct human use of the land. Consider the range of anthropogenic disturbances in a tropical forest, which includes agricultural deforestation, logging, fire, and fragmentation-induced edge effects. Selective logging degrades forests, resulting in local drying of these sites. Landscape fragmentation and land cover change interact synergistically to expose more of the forest to fire and consequently raise the risk of unintended fires occurring across the entire landscape (Veríssimo et al., 1995; Nepstad et al., 1999a; Cochrane 2001; Cochrane and Laurance, 2002).

Advances in understanding the fine-scale, but large-area, patterns through remote sensing are also making substantial contributions. Skole and Tucker (1993) and Woodcock et al. (2001) advanced a technique for fine resolution observations of the rate, pattern, and extent of forest cover change over large areas. Other observation efforts reveal the dynamics of regeneration (Alves and Skole, 1996; Steininger et al., 2001) and in contrasting studies, observations are also revealing areas with increasing woody vegetation, primarily in dryland ecosystems (Asner et al., 2003). Examples include the role of different resolutions of analysis on outcomes (Lambin and Strahler, 1994; Laris 2002; McConnell 2002), detection of cryptic deforestation and various stages of successional growth (Brondizio et al., 1996; Nepstad et al., 1999a; 1999b; Moran et al., 2000; Laris 2002; Batistella et al., 2003), improved fire detection (Laris 2002; Rogan and Franklin, 2002), and various regional and sectoral studies (Lambin 1997; Lambin and Ehrlich, 1997; Seto et al., 2000; Lupo et al., 2001). An extensive collection of references to these works is found in the various chapters in this volume.

Significant interest exists in understanding the drivers of land change, recognizing their complexity and variation, in order to improve its understanding beyond the broad factors of demand for resources from increasing population and levels of consumption. Significant headway has been made including the social causes of deforestation and arid land degradation (e.g., Moran 1993; Indrabudi et al., 1998; Robbins 1998; Sierra and Stallings 1998; Reynolds and Stafford Smith 2002; Walker et al., 1999; Archer 2003; Lambin et al., 2003); the role of institutions in land-use decisions (e.g., Lambin et al., 2001; Turner et al., 2001; Ostrom et al., 2002; Klooster 2003); and understanding the reciprocal relationships between population and land change (e.g., McCracken et al., 1999; Crews-Meyer 2001; Döös 2002). Significant gains have also been made in how to link social with physical processes using remotely sensed data and in nesting data and studies from local to regional to global scales (e.g., Moran and Brondizio, 2001; Fox et al., 2002; Walsh and Crews-Meyer, 2002; Turner et al., 2003), including the means of comparing different land classifications used in various studies (Gregorio and Jansen, 2000; McConnell and Moran, 2001).

Understanding of the role of population has also changed. From thinking that more people always meant less forest, a growing number of cases suggest that forests can persist under high population densities (e.g., Ostrom et al., 2002). The role of communities and institutionalized rules of management plays a critical role in such cases, emerging from a variety of sources, among them scarcity of the valued good (Turner, MD 1999; Laris 2002). Studies have shown how political and economic structures constrain individual choices about management of land resources (e.g., Robbins 1998; Archer 2003). Cultural traditions, and land tenure rules, are critical in influencing how land can be used and by whom (Tucker 1999). A notable advance has been the growing use of orbital earth-observing satellites linked to ground research to
address regional to local issues of land change (Liverman et al., 1998; Fox et al., 2002; Walsh and Crews-Meyer, 2002; Wood and Porro, 2002), contributing novel insights to the interpretation of land-cover change on topics rarely addressable with any accuracy at global or regional scales—e.g. land change in areas undergoing urbanization (Seto and Kaufman, 2003); and stages of secondary succession and their management (Brondizio et al., 1994; 1996; Moran et al., 2000).

3.3 LAND MODELING

An initial rationale for emphasizing land-change dynamics in global environment change science was to enhance earth system models. The modeling community, from economics to engineering has responded strongly to this element of land-change research. Significant advances are underway in a variety of modeling approaches, almost all of which focus on spatially explicit outcomes, aimed at explaining and projecting land-change (Lambin 1994; Rotmans and Dowlatabadi, 1998; Veldkamp and Lambin, 2001; Irwin and Geoghegan, 2002). Logit and other types of models explore the specific causes of land change drawing on various theories of the same (Chomitz and Gray, 1996; Pfaff 1999; Geoghegan et al., 2001; Vance and Geoghegan, 2002). Empirical models explore the robustness of land-cover change projections based on patterns of past change (Dale et al., 1994; Turner, M.G. et al., 1989; Turner, M. et al., 1989). Significant advances are underway in agent-based integrated assessment models in which the synergy between socially constrained human decision making and environment are linked to provide spatially explicit outcomes (Reibsame and Parton, 1994; Veldkamp and Fresco, 1996; Fischer and Sun, 2001; Parker et al., 2002; 2003). The range and amount of activity currently generated in the land modeling community is so large that it is better grasped by reviewing various sections of this volume. It is noteworthy, however, that the advances underway require new metrics by which to judge the results of the models. These, too, are being developed by the land modeling community (Pontius 2000; 2002).

4 The Future of Land-Change Research

The land-change programs worldwide continue to gain programmatic support as the magnitude, reach, and consequences of human-induced changes on the Earth’s terrestrial surface are understood, giving rise to such international science efforts as DIVERSITAS, the Millennium Ecosystem Assessment, and PLEC (People, Land Management, and Ecosystem Conservation—a United Nations University Project). These efforts and those such as the NASA-LCLUC program have launched integrated land-change science, demonstrating the significance and understanding gained from addressing cross-disciplinary problems of global environmental change, earth systems, sustainability, environment-development, conservation and countryside biogeography, among others (Turner 2002). We suspect that the new Land Project of the IGBP-IHDP will build on the breakthroughs of the past, and ensure the advancement of policy-relevant land-change science. This future, however, is predicated on the continuance of programs like NASA-LCLUC.

Several new initiatives promise to support this future beyond NASA and such programs, e.g. NSF’s Biocomplexity effort. As required by law, the USGCRP has been
formulating the Science Plan for the next decade of activities. Land-use and land-cover change appears to be a critical element in this developing program because of its stand-alone significance, and its critical role in other aspects of global change, in particular carbon cycle and ecosystem services research, and sustainability themes. The Climate Change Science Program has highlighted the critical importance of land-use and -cover change in setting new directions for global change research. The National Science Foundation is opening a new directorate-level program on Environmental Research in response to the “NSF Doubling Act” signed into law this year, which will significantly increase funding over the next five years for research linked to such themes as Complex Environmental Systems research, with a strong role for Coupled Human Natural Systems research, of which land use/cover change is key. Working with NASA and NOAA, the EPA (Environmental Protection Agency) will be putting in place new programs on regional-scale issues associated with global change. Development assistance agencies, such as USAID, are also building new programs around geographical information for sustainable development as a contribution to global change (NRC 2002), with land use and cover change research being highlighted as a core element of this agenda.

Emerging trends in science programming with respect to land-change studies are clear in their direction: increasing emphasis on place-based research, the science of forecasting, coupled human-natural systems, interdisciplinary research, and relevance to decision making. Building an agenda for global change at scales that matter, an emerging theme in global change and environmental research, calls for a strong role for an integrated science of land-use and -cover change.

5 References


DEVELOPMENT OF LUCC AND ITS LINK TO LCLUC


1 The NASA Programmatic Context for LCLUC Research

A program of Land Cover and Land Use Change (LCLUC) research is sponsored by the Earth Science Enterprise (ESE) within the National Aeronautics and Space Administration (NASA) (Asrar et al., 2001). The ESE’s research programs study the Earth as an integrated system, emphasizing observations made from the unique perspective of space, together with underlying laboratory, field, theoretical and modeling research (NASA 2003). The goal of ESE is to develop a scientific understanding of the Earth system in response to natural and human induced changes and improve predictive capabilities for climate, weather, and natural hazards. An understanding of land cover and land use change are essential for ESE to meet its science goal.

ESE’s strategic objective is to provide scientific answers to the overarching question of “How is the Earth changing and what are the consequences for life on Earth?” To address this overarching question, ESE identifies five fundamental scientific components for study: variability, forcing, responses, consequences and prediction. The LCLUC key science questions are: 1) Where are land cover and land use changing, what is the extent of the change and over what time scale? 2) What are the causes and the consequences of LCLUC? 3) What are the projected changes of LCLUC and their potential impacts? and 4) What are the impacts of climate variability and changes on LCLUC and what is the potential feedback? These questions are in keeping with the broader international research agenda on land-use and land-cover change (Moran et al., this volume).

LCLUC is a crosscutting science theme within the ESE designed and implemented initially by Robert Harriss and Anthony Janetos (the first LCLUC Program Manager). David Skole, chairman of IGBP/IHDP Land Cover and Land Use Change (LUCC) project and IGBP-DIS Focus 2 leader, and Chris Justice, IGBP-DIS Focus 1 leader, were instrumental in the emergence and early development of LCLUC at NASA through their involvement in broader national and international developments and their coordination of global satellite derived data sets for The International Geosphere-Biosphere Programme (IGBP).

Aspects of land-cover and land-use research are found throughout ESE sponsored projects and programs. LCLUC elements are found as part of ESE research in hydrology, ecology and biogeochemistry and in ESE programs including – the Interdisciplinary Science (IDS) Program, the data-oriented initiatives, such as the EOS Data Pathfinders (Maiden and Greco, 1994), and the Earth Science Applications
The LCLUC Program has a special place in NASA’s ESE, developing interdisciplinary science with a high degree of societal relevance, contributing to the US Climate Change Science Program (CCSP 2003). The LCLUC program aims to develop and use NASA remote sensing technologies to improve understanding of human interactions with the environment and, thus, provide a scientific foundation for understanding the sustainability, vulnerability and resilience of human land use and terrestrial ecosystems. In doing so, a major goal of the program is to further the understanding of the consequences of land-cover and land-use change on environmental goods and services, the carbon and water cycle and the management of natural resources (Janetos et al., 1996).

The longer-term objectives of the LCLUC program are to develop the capability to perform repeated global inventories of land use and land cover from space, to improve the scientific understanding of land-cover and land-use processes and models necessary to simulate the processes taking place from local to global scales, to model and forecast land-use and land-cover change and their direct and indirect impacts and evaluate the societal consequences of the observed and predicted changes. The NASA LCLUC research will also contribute to the U.S. Climate Change Research Initiative by establishing the operational provision of land-use and land-cover data and information products, services, models and tools for multiple users, e.g. scientists, resource managers, decision makers and policy makers (CCSP 2003).

2 LCLUC Research Components

The LCLUC program was designed initially around a number of regional case studies, complemented by methodological studies exploring the production and validation of particularly important regional and global remote sensing land use and land cover related datasets. The first phase of the LCLUC program (1996-2000) emphasized regional activities in areas where important land-use changes are taking place, or have recently taken place, such as the Brazilian Amazon, Mexico, Central and Southern Africa, China and Southeast Asia. The case studies used a combination of space observations, in-situ measurements, process studies and numerical modeling to address a combination of forcing factors of change involving climate, ecological and socioeconomic drivers, the processes of change, the responses and the consequence of change (e.g. Sader et al., this volume; Seto et al., this volume).

Understanding the processes of change is a prerequisite for predictive capabilities. Land use and land cover change is caused by a set of complex interactions between biophysical and socioeconomic variables. Understanding the process of land-
use change involves an understanding of human decision making. The regional case studies have been structured to strengthen interdisciplinary science, in most cases involving collaborations among ecologists, economists, geographers, remote sensing specialists and resource managers. In this sense, the LCLUC program has proven itself as a truly interdisciplinary research program within the ESE, forging partnerships between physical and social scientists to address questions critical to our understanding of the Earth System (see Section II of this book).

The biophysical and human forcing factors that result in changes of land cover and land use are manifested through different phenomena such as changes in land ownership or land management practices (e.g. urban expansion, agricultural expansion or abandonment, deforestation, logging, reforestation), and natural hazards (e.g. fires, droughts, floods, insect infestations) (Ojima et al., 1994). Variability in weather, climate, socio-economic conditions and internal ecosystem dynamics drive land cover changes on temporal scales from days (e.g. hurricanes) to years (e.g. logging) to decades (urban sprawl). Climatic and hydrologic variations and extremes can trigger persistent land cover changes that will, in turn, affect land-atmosphere exchanges for long periods of time (Avissar and Pielke, 1989; Pielke et al., 2002). Successive years of drought or above average rainfall, for example, can change land use practices or the frequency of fires (Dale 1997). Population changes and economic activity are critical factors that determine the type, distribution and intensity of land-cover modification and changes in land use (Marschner 1959; Meyer 1995). The different pressures of economic development around the world and the need for increased food production need to be expressed in quantitative terms and ultimately incorporated into predictive models of land-cover dynamics. These models need to be dynamic and process based, tuned to the forces of change at the regional scale (Desanker and Justice, 2001).

The consequences of land-cover change include a wide range of impacts on biogeochemical and water cycling, human health and welfare and biodiversity (Burke et al., 1991; Hansen et al., 2002). Measuring the rates of rapid conversion of forest cover to other types and monitoring the fate of deforested land is of particular interest because of the linkages to the carbon cycle, trace gas emissions, biodiversity, micro- and macro-climate, and sustainable development (Skole and Tucker, 1993; Houghton 1994; Turner et al., 1995; McGuire 2001). Another important topic is the consequence of intensified management in agriculture, forestry and grazing systems and assessment of land degradation processes in various ecosystems (e.g. Prince, this volume).

Because ecosystems often respond slowly, understanding current land-cover patterns and the present impacts of past land-use changes, requires taking into account land-use history. In this regard, the NASA LCLUC program contributed to funding an interagency effort led by US Geological Survey on Land Use History of America (Sisk 1998). There is a need for integration of multiple sources of land-cover data for longer historical perspectives. For instance, historical maps, based on accurate ground surveys, provide occasional, discrete records of land cover for the period 1850 to 1930; aerial orthophotos, acquired by various US agencies since 1930 and, since the mid–1980s, through the National Aircraft Photography Program (NAPP), a USGS led federal interagency initiative, can be interpreted for land cover for the period circa 1930 to the present; and unclassified Department of Defense satellite photographs provide imagery for the period 1960 to 1980. These sources may be combined with Landsat Multispectral Scanner System (MSS) and Thematic Mapper (TM) data for detailed retrospective studies covering the last 100 years.
The ESE LCLUC research contributes to a broader effort by the US government research agencies to incorporate human dimensions into the study of environmental changes, and for the main part, NASA relies on partnerships with other agencies, such as the National Science Foundation (NSF) and National Institute of Health (NIH) for the development of historical and socioeconomic data sets and methods and models for analysis of these data (National Research Council 2001).

The emphasis of the current phase (2001-2004) in the NASA LCLUC Program is on extending local process and case studies to the regional scale and undertaking synthesis efforts across world regions; studying the impacts of land-use change with emphasis on carbon and hydrological cycles; and predictive modeling of future land use, as well as on developing improved regional data sets; field process and parameterization studies; and improved modeling of land-use dynamics.

The scaling up of land-cover and land-use change studies underscores the importance of ready access to data sets of increasing size and complexity. Since its inception, the LCLUC program has fostered better access to NASA Earth observations and modeling results related to land processes. An important component of the LCLUC program is the compilation and distribution to the scientific community and the public of large, remote sensing-based data sets related to land-cover and land-use change. Recent examples of such data sets include the orthorectified, global Landsat data sets compiled by Earth Satellite Corporation from Landsat data from circa 1990 and 2000 (Sheffner et al., 2003) and the MODIS Land Product suite (Justice et al., 2002).

3 Remote Sensing and the Study of LCLUC

Spatially explicit monitoring of changes in land cover is essential for LCLUC research and is now feasible from space (e.g. Skole and Tucker, 1993). The foremost observational requirement for LCLUC research is to extend the 30-year record of high-resolution Landsat observations, which constitutes the indispensable foundation for global land cover inventories. Availability of the global 30-m spatial resolution data that have been recently collected from Landsat-7 increases the potential for detailed studies of landscape change, a task that only a few years ago was not possible for some regions of the world (Arvidson et al., 2001; Goward et al., 2001). At the time of writing, Landsat 7 is malfunctioning and there is considerable uncertainty concerning the continuity of the data record. It is critical that the long-term record of high resolution observations from the Landsat program be continued. For any follow-on mission, it is important that the data quality and availability will be adequate for LCLUC research. To achieve this, emphasis will need to be given to instrument calibration, data quality assessment, data acquisition strategy and data policy. It is now more critical than ever, that high-resolution observations (c. 30m) become part of the operational environmental satellite suite, enabling data continuity without gaps in the data record (Committee on Earth Sciences 2000).

High-resolution satellite data are used to map and quantify land cover changes at the local to regional scale; moderate resolution data are used to classify and characterize land cover at the global scale (Estes et al., 1999). This combination of moderate and high resolution satellite remote sensing forms the basis for a land cover change observing system (Skole et al., 1997). The Terra and Aqua satellite products, from the Moderate-resolution Imaging Spectroradiometer (MODIS), extend the
moderate resolution data record from the Advanced Very High Resolution Radiometer (AVHRR), providing global land-cover products, including continuous fields of tree cover and burned area (Friedl et al., 2002; Hansen et al., 2002; Justice et al., 2002; Townshend and Justice, 2002). A new development led by the MODIS science team has given increasing emphasis to data product quality assessment and validation, which sets an important precedence for future missions (Morisette et al., 2002; Roy et al., 2002). The unprecedented 250-m spatial resolution of MODIS daily global observations is being used to accurately detect changes in land cover, allowing the monitoring of potential “hot spots” over the globe (Zhan et al., 2002). The continuity of these moderate resolution data is being developed in the context of the National Polar Orbiting Environmental Satellite System (NPOESS) (Townshend and Justice, 2003). The data from the next generation moderate-resolution sensor -- the Visible Infrared Imaging Radiometer System (VIIRS) -- will need to be of sufficient quality to enable the measurement suite developed for the MODIS time series to be continued.

The NASA ESE and its LCLUC Program support taking data from the sequence of individual moderate and high resolution sensors and developing long-term climate data records of sufficient quality for the study of global change. The challenge in creating these records is to account for the evolving instrument design and performance since the 1970’s and necessitates dynamic product continuity.

During the last few years the LCLUC program has been investigating the utility of hyperspatial resolution data (<10m) for mapping land-cover change, using data from the commercially operated IKONOS system. Results have demonstrated the utility of hyperspatial data for mapping logging and forest degradation and validating high-resolution interpretations (Hansen et al., 2002; Laporte et al., this volume). NASA has also launched technology demonstration projects to explore the utility of high resolution thermal and hyperspectral data, such as the New Millenium EO-1 mission (Yamaguchi et al., 1998; Ungar et al., 2003).

Priority for change detection is being given to areas of the world that have been undergoing the most change and where stresses from human activities are likely to increase most rapidly. Emphasis is placed on the exploitation of satellite remote sensing data as the best source of information on the spatial distribution of land cover and rates of landscape change on regional, continental, and global scales. Time series of satellite data are analyzed to provide a consistent record of global land-cover change, characterize the end-states of land-cover modification in regions of high population density, and study the impacts of spatial patterns and past history of land-use conversion on ecosystem processes, biodiversity, and the sustainability of ecological services, such as recycling of nutrients.

4 LCLUC and ESE Applications

In addition to the scientific research programs, the LCLUC program is developing a partnership with the ESE Earth Science Applications Division to assure that the Earth observations, modeling and system engineering it supports to acquire land cover/land use information are extended for use by the broader community for decision support in applications that address issues of national importance. The Earth Science Applications Division is identifying decision support systems used by NASA’s operational partners to benchmark existing technology in the procedures of the partner organizations, and to
improve the operational efficiency of the partners through incorporation of NASA’s unique assets and capabilities. Land cover and land use information is crucial in most of the twelve national application areas identified to date by the ESE. The role of LCLUC in four of these application areas is described below.

4.1 LCLUC AND CARBON MANAGEMENT

Carbon management is a critical national issue. Within the next several years, there will be a need for information on carbon exchange, both domestically and internationally. Land-cover and land-use information will be needed to estimate the potential of a given area as a source or sink of carbon, monitoring land-cover and land-use change for carbon sequestration and assessment of carbon sequestration projects (IPCC 2000; Marland et al., 2003). The primary data need will be rigorous, defensible, spatially explicit estimates of carbon exchanges at spatial and temporal scales that match the size of land being managed and the temporal constraints of the carbon credit exchanges and contracts.

Users will likely need to have uncertainty estimates that incorporate the various potential sources of error in input variables, model relationships, and parameter estimates, as they sum over the time and spatial extent of the estimates. Much of the activity in carbon trading will involve individual plots of land or management operations. Consequently, research is needed to develop methods and techniques at fine spatial scales. The focus of this research must include evaluation of sources of input data, and methods for fusing data from different sources, as well as how to represent landscape level variability in simulation models. Attention will be needed in those areas with greatest carbon sensitivity, for example: 1) where dramatic land-use change is occurring, in combination with either 2) areas with very high flux rates of carbon (high NPP, any wet and warm ecosystem), or 3) areas with very large storage of carbon (e.g. high amounts of carbon stored in soils or vegetation, such as peat lands or the tropical forests).

4.2 LCLUC AND WATER MANAGEMENT

Land-cover and land-use information is essential in decision support systems for water management and includes monitoring parameters affecting water quantity and quality and generation of data products that characterize riparian zones, land cover type and vegetation structure and condition. Land-cover information combined with digital topography will provide significant new data products for hydrologic modeling. High resolution digital elevation models combined with high resolution imagery of flood plains will assist the Federal Emergency Management Agency (FEMA) and local government agencies plan for floods and respond to flood events, with more accurate information on the expected rate and extent of flooding. NASA Shuttle Radar Topography Mission (SRTM) digital elevation data will improve maps of snow pack and provide more accurate estimates of amount and impact of runoff.

The utility of geospatial information for decision support systems is dependent, in part, on the spatial and temporal resolution of the data. For example, for water management, a spatial resolution of at least 1 km on a weekly basis is “reasonable” for snow pack estimation. Monthly coverage at the same spatial resolution or higher (e.g. MODIS 500m) is suitable for vegetation monitoring or burn
scar mapping, 30m data within a few days after the burn are needed for post-fire
watershed rehabilitation assessments. Regional thirty-meter resolution data (e.g.,
Landsat ETM+) could be updated semi-annually to annually for increased benefit and
improvement in decision support systems. Incorporating higher resolution datasets into
end-user needs would be beneficial in both the immediate future and for years to come.

4.3 LCLUC AND COMMUNITY GROWTH

The issue of scale is critical when discussing geospatial information needs for
community growth and disaster management issues. While much of NASA’s
capabilities in earth observation, modeling and systems engineering have been aimed at
regional to global consideration, community growth and disaster management issues
are usually focused on local areas and larger scales. The challenge to the research and
applications community is to adapt the techniques and the technology of geospatial
analysis to the local level. Depicting the context of human/land interface is the
dominant geospatial information requirement for community growth and disaster
management. Urban vegetation inventory, impervious surface mapping, transportation
planning and networking, land-cover impact on water quality/quantity, and planning for
and response to natural and man-made disasters – wildfire, flood, earthquake and severe
weather - are specific application areas. Regular mapping of the urban interface needs
to be undertaken using high-resolution satellite or airborne data. The use of cellular
automata and agent-based models are being used experimentally in combination with
Landsat data to develop projections of urban growth (Clarke et al., 1997). Local
government communities base decisions on high spatial resolution information. This
information is often described in terms of parcels. In addition, implications of change
are measured in days, not years. Thus, data for local decision-making must maintain
high spatial and temporal resolutions.

While hyperspatial resolution data (< 10m) are becoming available for local
uses, there remain significant hurdles to adoption including access to data and
knowledge of the use of geospatial information (Sugumaran et al., 2002). Airborne
Lidar technology is being used increasingly for fine scale mapping of everything from
ecosystem functioning to urban areas providing canopy structure and elevation data that
can be applied directly to existing/traditional applications (Lefsky et al., 2002; Palmer
and Shan, 2002). Beyond limitations in spatial resolution, land-cover and land-use
applications need to address the scaling of models to local communities.

4.4 LCLUC AND INVASIVE SPECIES

Invasive species are a major environmental issue of this century and have recently been
identified as one of the eight grand challenges in environmental science (National
Research Council 2001). The number of invasive species entering the US per year is in
the thousands (Simberloff 1996). This issue has developed diverse stakeholder support,
ranging from land management agencies, states, the agricultural industry, conservation
organizations, and private landowner groups.

The invasive species application includes two broad issues, mapping location of
known and potential areas of invasive species expansion, and location and eradication of
invasive species. Common to both issues is the need for rapid input and processing of
data and the ability of local managers to acquire the geospatial information they need to
take effective action. Invasive species, once well established in an area, are virtually impossible to eradicate for biological and financial reasons. Current land-cover and land-use information alone is not sufficient for predictive mapping. Land cover maps derived from remote sensing need to be designed specifically to meet the needs for invasive species studies and merged with other geospatial data pertinent to the spread of invasive species (e.g. topography, microclimate, land use). Habitat and species information is crucial for invasive species control.

Hyperspectral remote sensing is being evaluated for its use in invasive species mapping (Underwood et al., 2003). Existing programs focus on development of predictive maps of the spread of invasive species and keeping those maps current with information from national and local organizations. Control efforts require the integration of ground data with local to regional geospatial data products of land cover, topography, soils, water resources, weather and climate. Effective prediction of invasive species habitat and spread requires technology that is beyond the capacity of local communities and must be delivered through cooperative and collaborative efforts. The use of the Internet provides a mechanism for quick assimilation of data, data processing, product generation and product delivery.

5 US Interagency Coordination in the Study of LCLUC

NASA ESE is an active contributor to the US Global Change Research Program (USGCRP), which is the interagency forum for coordinating global change research. In the first decade of USGCRP, emphasis was given to addressing climate change causes and impacts (Committee on Global Change Research 1999). For the second decade of the USGCRP, Land Use and Land Cover Change has been added as a research element, and a Climate Change Research Initiative (CCRI) has also been included to address those research aspects of immediate societal relevance (USGCRP 2002; CCSP 2003). In this context, partnerships are now being developed with other federal agencies such as USGS, USDA, USFS, EPA, NIH and USAID to expand the scope of the land-use and land-cover change research agenda. Strengthening interagency collaboration to complement NASA’s research is considered a top priority of the NASA LCLUC program.

Another important development in the United States has been the identification in 2001 of Land-Use and Land-Cover Change Dynamics as one of the eight top challenges in the environmental sciences (NRC 2001). The discussion by a blue-ribbon panel of the NRC considered more than 200 topics before focusing on 8 of them as the Grand Challenges for the coming decade. Land-use and land-cover change was identified as a fast-track priority requiring immediate funding because of its significance and the readiness of the community to make important contributions given the efforts of the past decade.

6 The International Context of the LCLUC Program

Important land-use questions related to global food supply, regional water resources, carbon sources and sinks, biodiversity loss and population growth necessitate working in a number of different regions and results in a strong international flavor to the
LCLUC program. The program emphasizes studies where land-use change is rapid or where there are significant regional or global implications. The LCLUC program promotes collaboration with in-country scientists and regional science networks, increasing their accessibility to NASA spaceborne assets and helping NASA scientists analyze and understand the often complex local and regional issues and conditions. In addition to individual regional research projects, the ESE has recently supported two large regional science campaigns with an emphasis on LCLUC: the LBA project studying land use in the Amazon Basin incorporate the study of LCLUC in the Amazon Basin (Nobre et al., 2001) and the SAFARI project studying fire and its impact on atmospheric chemistry in southern Africa (e.g. Korontzi et al., 2003).

The IGBP-IHDP Land Use and Cover Change (LUCC) Research Program, which preceded the NASA program is described by Turner et al. (in this volume). Communication between LUCC and LCLUC has been routine, usually conducted through the informal channels of scientific exchanges and common participation in a variety of research activities. A number of LCLUC projects have been endorsed by the LUCC Program. Recently, LCLUC and LUCC have jointly established an international colloquium series consisting of periodic conferences on topics of mutual interest to both programs.

The LCLUC program recognizes the importance of establishing the global observing systems needed for the long-term monitoring of land cover and land use and through its research program is developing and demonstrating the methods required for such a system. NASA primarily develops and tests aerospace technology. The ESE develops and launches the instruments, generates the data products and develops the associated modeling and analysis methods to address earth science questions. Although NASA collects systematic observations in support of its science mission, it is not an operational agency and responsibility for operational monitoring within the US falls to other agencies e.g. NOAA, USGS, USFS (National Research Council 2003). It should be noted that the latter two agencies have a strong domestic emphasis and responsibility for international operational monitoring must ultimately be addressed in partnership with the international community. The global monitoring systems will need to be international, conforming to internationally accepted standards of data quality, product accuracy and data continuity.

Although global monitoring systems are rapidly being established for the global oceans (National Research Council 1997; IOC 1998), they do not yet exist for the land surface. In this regard the LCLUC program is a major contributor to the Global Observation of Forest Cover/Global Observation of Landcover Dynamics (GOFC/GOLD) program, which is a component of the Global Terrestrial Observing System (GTOS) (Justice et al., 2003; Townshend et al., this volume). This program has the goal of establishing the operational monitoring systems through international cooperation.

7 LCLUC Program Directions

Although the base funding of the LCLUC program has remained at the same level, during the last 3 years the LCLUC program has grown and the number of projects across ESE with LCLUC elements has almost doubled, mainly due to funded projects under NASA’s Carbon Cycle, Interdisciplinary and Landsat programs. The LCLUC
program has expanded from the initial LCLUC case studies to include studies on the impacts of land use and regional LCLUC prediction and synthesis. Currently, the program is comprised largely of projects studying the LCLUC relations to the carbon cycle and water cycle. The program’s strategy continues to be based on balancing large-scale satellite data analysis and generation of validated data sets with regional case studies designed to gain insights into specific biogeophysical and anthropogenic processes and their social contexts. Large-scale climate modeling with global or regional LCLUC satellite-derived and in situ data used as input is gradually becoming an important part of the program.

In the coming years the LCLUC program will emphasize its contribution the US CCSP, leading to projections of future land-use change and undertaking LCLUC studies to address the impacts of LCLUC on the carbon cycle, water resources and climate. The LCLUC program will continue its contribution to GOFC/GOLD, helping to secure regular global inventories of land-cover and land-use change. The LCLUC program will continue its regional process studies but also contribute to emerging regional science initiatives such as the North American Carbon program (NACP) and the Northern Eurasian Earth Science Partnership Initiative (NEESPI).

Many challenges remain for phase II of LCLUC: regional syntheses, advances in integrated modeling (Veldkamp and Lambin, 2001; Parker et al., 2002; 2003), applications of societal relevance, and further strengthening the link between the natural and the social sciences to ensure the integrative study of land change (Moran and Ostrom, under review). The synergistic use of remote sensing and social science data and enhancement of the interdisciplinary character of the NASA LCLUC program will remain the main programmatic goal.

8 References


Moran E.F., Skole D.S., and Turner B.L., II. The development of the international land use and land cover change research program and its links to NASA’s land cover and land use change initiative. - In this volume.


Seto K.C., Woodcock C.E., Kaufmann R.K. Land-Cover/Land-Use Change: Case Study of the Pearl River Delta, China – in this volume.


CHAPTER 3

MEETING THE GOALS OF GOFC
An Evaluation of Progress and Steps for the Future

JOHN R. TOWNSHEND¹, CHRISTOPHER O. JUSTICE¹, DAVID L. SKOLE², ALAN BELWARD³, ANTHONY JANETOS⁴, IWAN GUNAWAN⁵, JOHANN GEORGE GOLDAMMER⁶, BRYAN LEE⁷

¹University of Maryland, Department of Geography, 2181 LeFrak Hall, College Park, MD 20742 USA
²Michigan State University, Lansing, MI USA
³Joint Research Center, Ispra, Italy
⁴The H. John Heinz III Center for Science, Economics, and the Environment, Washington DC, USA
⁵Agency for the Assessment and Application of Technology (BPPT), Bogor, Indonesia
⁶University of Freiberg, Germany
⁷Canadian Forest Service, Edmonton, Canada

1 Introduction

To ensure systematic long-term observations of the environment several efforts have been made in recent years to provide international coordination of observational systems working within the framework of various overarching international organizations. One such effort is known as Global Observations of Forest Cover/Global Observations of Land Cover Dynamics (GOFC/GOLD). Originally it was set up to consider the monitoring of forests and hence the original name of Global Observations of Forest Cover (GOFC) but has been extended to all land cover types (see section 4.3).

GOFC/GOLD is one of the Panels of the Global Terrestrial Observing System, an organization sponsored by four UN bodies, namely the Food and Agricultural Organization (FAO), the United Nations Educational and Cultural Organization (UNESCO), the United Nations Environmental Programme (UNEP), the World Meteorological Organization (WMO) and the International Council for Science (ICSU).

GOFC as an organization was originally set up by the Committee for Earth Observing Satellites (CEOS) and it retains close links with that organization because of the importance of spaceborne remotely sensed observations to the goals of GOFC/GOLD. In this chapter we review the goals of the organization and make an assessment of progress made in realizing the plans formulated in the later part of the 1990’s. In doing the latter we propose a new template which is used to assess objectively how close we are to achieving the goals of long term operational systematic observations of land cover for the whole globe.
2 The International Roles of GOFC/GOLD

International organizations concerned with observations have key roles in ensuring that consistent global observations are collected. It has to be recognized that the resources that they are able directly to control are usually quite limited, most of the resources remaining at a national level. For example WMO helps coordinate an enormous array of *in situ* and space observation capabilities primarily for weather forecasting, but it directly controls only a quite modest budget. Even though most international bodies have limited resources they do play key roles in providing overall strategies and frameworks, setting well articulated designs and goals for global and regional observing systems and in establishing and gaining international consensus on standards and protocols relating to the collection of observations and the supply of data and information products. Additionally they have important roles in assessing whether observational systems are indeed meeting the goals that have been agreed upon by governments. Most such organizations also play an important role in capacity building in developing countries.

International organizations are usually less effective at directly raising resources themselves to carry out research, to collect observations or generate products: rather they provide the international framework within which national resources can be allocated. Although international organizations are relatively impecunious by design there are major exceptions where groups of nations decide to pool resources to carry out specific tasks. This is especially true within Europe, for example where Eumetsat is responsible for operational European weather forecasting satellites and the European Space Agency is responsible for environmental satellites for research purposes.

3 Origins of GOFC/GOLD

The design of GOFC/GOLD arose from substantial earlier efforts to monitor the earth’s land cover and especially its forests. The following are some of the important precursor activities identified in the original strategy document for GOFC prepared for CEOS (Janetos and Ahern, 1997). The remainder of this section is based closely on this document.

The World Forest Watch Conference held in São José dos Campos, Brazil (June 1992) provided an initial high-level international forum for the assessment of current approaches to satellite-based monitoring. This meeting also served as a basis for forwarding recommendations from the technical and scientific communities to the policy makers and government leaders at UNCED, and the international global change research community through IGBP. The conference concluded that significant technical and methodological advances have been made in recent years, and they were sufficient to proceed with an observation system which could satisfy both scientific and national-level forest management requirements.

NASA, in conjunction with the US-EPA and USGS, undertook a prototype procedure for using large amounts of high resolution satellite imagery to map the rate of tropical deforestation, one of the most important land cover changes. This activity, called the Landsat Pathfinder Project built on experience gained during a proof-of-concept exercise as part of NASA’s contribution to the International Space Year/World
MEETING THE GOALS OF GOFC

Forest Watch Project (Skole and Tucker, 1993) (Townshend et al., 1995). It focused initially on the Brazilian Amazon, and was expanded as part of NASA's Earth Observing System activities to cover other regions of the humid tropical forests. In principle, it provided an initial large-scale prototype of an operation program.

The TROPical Ecosystem Environment Observations by Satellites (TREES) project was implemented as a demonstration of the feasibility of applying space observation techniques to monitoring of land cover and biomass burning (Malingreau et al., 1993). This project utilized global coverage with a wide range of sensors including AVHRR, ERS-SAR, JERS-1 SAR, ATSR, Resurs, Landsat, and SPOT. It also focused on the use of thermal sensors for the detection of fires, and incorporated other indicators of deforestation.

In conjunction with NOAA and the USGS, NASA supported an IGBP project to acquire global, daily coverage with the AVHRR sensor at 1 km resolution (Eidenshink and Faundeen, 1994; Townshend et al., 1994). This project was the first of its kind to acquire global, daily coverage at this resolution. The resultant dataset was processed at the EROS Data Center and provided the basis for globally mapping land cover, vegetation phenology and fires. As part of a land-cover mapping exercise, this project eventually led to the DISCover global land cover data set (Belward et al., 1999; Loveland et al., 2000).

The FAO Forest Resource Assessment for the 1990 period (FRA-90) used a combination of earth observation data and national statistics to derive statistics about forest cover for the 1990 datum, and forest cover change for the period 1980 to 1990 (FAO 1995). For tropical forests, FAO developed a technique called the Interdependent Interpretation Procedure, which relied primarily on a two-date interpretation of large scale photographic prints of full Landsat scenes. Using training centers in Latin America, Africa, and Asia, FAO was able to train local interpreters and obtain consistent results. The resulting change classes also allowed FAO to estimate transitions between several woody biomass categories for Africa, Latin America and Asia (FAO 1996). Subsequently the FRA 2000 assessment has been reported (FAO 2001), though with concern expressed concerning its accuracy (Matthews 2001).

An early major example of a regional /continental scale land cover mapping project was the 1km AVHRR data set generated by (Loveland et al., 1991). A similar effort was undertaken in Mexico with the help of in-country collaborators. An Interagency Multi-Resolution Land Characterization (MRLC) project, implemented by the USGS on behalf of a number of federal agencies, subsequently generated land cover data sets for the conterminous US. This data set included products from coarse and high spatial resolution satellite sensors. This project also provided complete coverage of the United States with Landsat Thematic Mapper data.

Within the framework of NASA's Global Forest Mapping Program, the Global Rain Forest Mapping project was a collaboration between NASA, JPL and the Joint Research Centre with the aim of generating JERS-1 L-band SAR data sets of the global tropical forests. A complete SAR coverage over the tropical belt - in total some 13,000 scenes - was acquired in 1995-96, including multi-temporal coverages over the Amazon and Congo river basins to capture season effects and to map inundation in flooded forests. SAR mosaics at 100 m resolution were generated to cover the tropical region in all three continents. The mosaics were radiometrically and geometrically corrected, though not for topographical effects.
Starting in 1997, the Global Boreal Forest Mapping Project (GBFM) formed a follow-on project to the GRFM. The GBFM project aimed at covering the boreal forests in Siberia, Northern Europe and North America with JERS-1 SAR. Also the GBFM project is a joint effort by NASDA, NASA/JPL/ASF and the JRC, with collaboration from DLR, the National Swedish Space Board, CCRS, and CSA.

The Canadian Radarsat program acquired complete global coverage in the ScanSAR mode during its initial phase of operations (Mahmood et al., 1998). An associated research program was initiated to demonstrate applications, including applications to forest are research. Research showed that Radarsat held promise for mapping logging and deforestation in tropical and boreal forests (Ahern et al., 1997; Kux et al., 1997; Shimabukuro et al., 1997) and for monitoring macrophyte growth in tropical reservoirs and floodplains (Costa et al., 1997).

Many regional land cover mapping activities were initiated in the 1990s. Within the IGBP System for Analysis, Research and Training (START) a number of regional activities focusing on the land cover change question were begun. Most notable were the Southeast Asian and Southern Africa regional activities (IGBP 1997). These various initiatives led to the formulation of an overall observational strategy for the monitoring of land cover change (Skole et al., 1997), which contains many of the elements of the subsequent GOFC/GOLD programs.

4 Review of the Strategy of GOFC/GOLD

The strategic goals of GOFC/GOLD are to be found mainly in the original strategic plan written for CEOS in 1997 (Janetos and Ahern, 1997), in the more detailed documentation of the fine resolution (Skole et al., 1998) and coarse resolution remotely sensed products (Loveland et al., 1998) and subsequently in the revised integrated strategy prepared in 1999 (Ahern et al., 1999). The goals for the fire element were developed following the integrated strategy (Ahern et al., 2001; Justice et al., 2003).

4.1 PRIMARY STRATEGIC GOALS

The ultimate objective of GOFC/GOLD is to lead to the operational collection of sustained systematic observations and their processing to create useful products for several key user communities (table 1). The observations include all those required whether collected in-situ or from remotely sensed platforms of various types. Operational in this context means that nations will take on the responsibility for these activities on an indefinite basis just as they do for observations leading to weather forecasts.

The essence of the implementation strategy was to develop and demonstrate operational forest monitoring at regional and global scales by developing prototype projects along the three primary themes of Forest Cover Characteristics and Changes, Forest Fire Monitoring and Mapping and Forest Biophysical Processes. With the broadening of the scope of GOFC to include all land cover the term “forest” has been replaced by “land”.

As part of this process, teams have been assembled to guide the implementation, executing prototype projects, developing consensus algorithms and standard methodologies for product generation and product validation in conjunction
with *in-situ* measurements, and developing and demonstrating procedures for improved data access for the user community. Up to the present the main focus has been on the first two themes. An important part of implementation strategy has been to identify gaps and overlaps in earth observation data, ground systems, methods, and scientific knowledge from the experience gained in developing and executing GOFC prototype projects.

Table 1. Main user communities for forest and other land cover information
(adapted from (Janetos and Ahern, 1997))

| (1) Global Change. | There are three pathways of relevance for global change and forest lands with implications. (Turner II et al. 1995). The first is the interaction of forests and the atmosphere: regulation of the hydrologic cycle and energy budget, with implications for weather and climate prediction. Understanding these interactions is a crucial part of the global change science agenda. The second is both a scientific and policy issue: the implications of changes in forest land for the atmospheric carbon dioxide budget. Are forested lands net sources of carbon or net sinks of carbon? What are the rates of deforestation and reforestation? What are the implications of changes in fire frequency? The third pathway is the potential impacts of climate change: what effect will changes in temperature, precipitation, and concentrations of carbon dioxide and other radiatively active gases have on forest composition, productivity, health, and distribution, and therefore on the economic activities that are generated by forested lands? What feedbacks might these changes have to the climate system itself? |
| (2) Timber, fuel, and fibre. | For centuries, human society has depended critically on the continued supply of wood and fiber from forest land. Many societies have declined after depleting their forest resources (Perlin 1991). The twentieth century produced substantial improvements in the efficiency of production of wood and fiber from forests. Increasing populations and increasing literacy and affluence in the twenty-first century will continue to put increasing demands on the forests of the world. Continued improvements in forest management will be necessary to meet these demands without depleting the forest resource (Nyland 1996) |
| (3) Watershed Protection. | Forested areas and other wooded lands play a crucial role for terrestrial hydrological systems (Hammond 1991; Nyland 1996). These areas constitute the most important buffer zones for both runoff and infiltration to ground water reserves. The actual location and size of forests in relation to the watershed often determines the run-off fraction of the precipitation and becomes the dominant factor in flood and erosion prevention. Close monitoring of forests in relation to the hydrographic network thus becomes a critical issue in physical planning, landscape planning, and environmental protection in all regions of the world. |
| (4) Biodiversity. | Concern over the rapid, human-driven loss of biodiversity worldwide is rapidly mounting, culminating in the International Convention on Biological Diversity. The extent and condition of forested lands are central issues to the preservation and sustainable use of biodiversity. Both forest loss and forest fragmentation have been implicated in losses of biodiversity by the scientific community. On the other hand, many national programs for sustainable forest |
management are now incorporating biodiversity as one of the attributes to be enhanced and maintained.

(5) Recreation and Tourism. Forests provide habitat for wildlife, and opportunities for recreation and tourism. In many countries, the direct economic return from these activities approaches the return from extraction of wood and fibre. Many other countries want to increase the benefits they derive from recreation and tourism. Information obtained from Earth observation data can aid in the strategic planning for recreation and tourism in forested areas.

(6) Sustainable Forest Management. Sustainable forest management has become an extremely important philosophy, along with ecosystem management, to guide the plans of land-managers (both public and private) for the use of forested lands over the next several decades. Sustainable forest management recognizes the multiple benefits provided by forests, and strives to respond to the needs and wants of the multiple stakeholders who benefit from the forest, while conserving its resources so future generations can also benefit from the forest. Earth observation data, in conjunction with geographic information systems, provides exciting new opportunities to engage a broad spectrum of stakeholders in long-term forest management planning.

GOFC/GOLD is being implemented through a series of regional networks, providing a forum for regional scientists, data providers and operational users to articulate their information requirements and improve access to and use of the observations (Justice et al., 1999). These regional networks also provide a mechanism for calibrating, validating and improving methods and algorithms and a place to test integration of in-situ and remote sensing observations.

4.2 SUBSIDIARY STRATEGIC PRIORITIES

In order to achieve the first order strategic goals, a series of other subsidiary goals were also identified in the original plans of GOFC/GOLD, especially that it should work towards the increased operational use of earth-observation data for policy decision-making at national, regional, and global levels. Validated products should be generated, which can be used to derive credible information concerning the forest and other land cover types’ components of the carbon budget for research and policy use. Using as appropriate, data from multiple sensors, in combination with in-situ data, validated prototype information products should be created, which satisfy clearly identified requirements of user agencies. The use of earth-observation information products for forest management and scientific research concerning forest biophysical processes also needs to be enhanced.

Promotion of common data processing standards and interpretation methods, necessary for inter-comparison of regional studies should be pursued. There should be stimulation of advances in the state of the art in the management and dissemination of large volume data sets and information from multiple sensors. Partnerships between CEOS space agency members and user agencies should be created and strengthened and deficiencies identified, as well and recommendations about how unnecessary overlaps in space agency programs might be resolved.
4.3 ARE THE PRIORITIES FOR GOFC/GOLD STILL SOUND?

Consideration of the plans as outlined in the previous two sections for GOFC/GOLD it is fair to conclude that they remain sound and timely. The fact that they remain timely should not be construed as meaning that no progress has as yet been made (see section 6). The most important change was the inclusion of all cover types and not merely forest cover. The reasons for this partly lie with the quite broad definitions of forest cover and was particularly related to the official United Nations Forest Resource Assessment definition. The definition of forests is based on a number of factors including crown cover which need only be as little as 10% (FAO 2001). Therefore a substantial proportion of all the earth’s vegetated areas are likely embraced by this definition and in physiognomic terms only grasslands with less than 10% crown cover and unvegetated areas would be excluded. Unless GOFC/GOLD includes the other cover types in its remit it would leave the necessity for another body to take responsibility for this component of the Earth’s land cover types. Also encouraging an all embracing approach to the earth’s land cover by GOFC/GOLD is the increasing interest of forest services in many areas of trees in non-forested areas. For these reasons GOFC/GOLD now includes in its remit all land cover types including forest cover.

It should be stressed that these considerations do not mean that the definition of forests for all GOFC/GOLD’s stake-holders exactly match those of the FAO Forest Resource Assessment. The latter also requires that the dominant land use is also forestry. Hence an area with crown cover is >10% where the dominant land use is for livestock production is not a forest sensu stricto. For many of the stake-holders of GOFC/GOLD observations of forest cover related to their biophysical properties are observed independent of land use (see table 1).

Consideration of the all-embracing original goals of GOFC/GOLD indicates they have been somewhat overly ambitious especially given the funds that have become available. This is manifested for example in the fact that three implementation teams were originally proposed namely forest (land) cover, fire and biophysical. In practice only the first two have been set up and the biophysical team still had not been convened. In part this relates to lack of resources but in part it also relates to the fact that other international efforts have recently been making major advances in either scoping and defining the needed observations such as the activity known as Terrestrial Carbon Observations (TCO) (Cihlar et al., 2002) (Cihlar and Jansen, 2001) or in the actual implementation such as the Fluxnet programs (Baldocchi et al., 2001). Clearly it is inappropriate that GOFC/GOLD attempt any sort of duplicative international coordination role. However, consideration of these activity’s plans indicates the clear need for GOFC/GOLD in terms of the required remotely sensed observations.

A third deficiency in the plans of GOFC/GOLD is the absence of a set of metrics for measuring the progress made in achieving its stated goals in implementing its plans. Without such metrics any organization will lack the necessary mechanisms for monitoring its progress and for, as needed, revising its approach. In the next section we outline a template for implementation of GOFC/GOLD and discuss each of the stages in turn, in terms of what is required within each. This approach is based on achieving the three main goals of GOFC/GOLD namely making observation systems operational, making products more available and ensuring that the product are
gainfully applied by stakeholders. Following this section is an assessment of current progress made in each of these.

5 A Template for GOFC/GOLD in Operationalizing Products

To move from a set of plans to the operational collection of a set of systematic long-term observations is always very challenging however important the observations (Karl 1996). In Figure 1 we provide a template showing the main stages which have to be gone through. It should be noted that the process is almost always an iterative one. Once observational products are created and used on a regular basis they are rarely optimal for all users, nor will requirements normally remain static. It also needs to be recognized that in achieving such goals, implementation usually remains largely with specific programs in national agencies rather than with resources channeled through international organizations. Thus international observing programs need both formal and informal mechanisms to influence national agencies in terms of implementation. For example, currently the operation of all environmental satellites remains with individual space agencies. One notable exception to this is Eumetsat, which acts on behalf of Europe’s meteorological agencies and has a governing council of the member nations. The difficulties in trying to influence several different space agencies is in many respects quite minor compared with getting national agencies to become responsible for in situ observations.

Template for GOFC/GOLD to operationalize its products

![Diagram showing the main stages in the development of capabilities within the framework of GOFC/GOLD]

Figure 1. Template of the main stages in the development of capabilities within the framework of GOFC/GOLD

Overarching international coordination mechanisms such as the Integrated Global Observation Strategy and the Strategic Implementation Team of CEOS offer possibilities for improving observational systems as does the recently launched GEO coordination scheme following the Washington DC summit of environmental ministers.
in 2003. Whatever the mechanisms used to achieve implementation there needs to be clarity in measuring progress for each major set of observations.

5.1 SPECIFICATION OF REQUIREMENTS

The requirements for products need to be clearly stated in quantitative terms. They should arise from wide consultation with user communities and they should be grounded in the refereed scientific literature so far as possible. It is also desirable that the consequences of not achieving a particular goal be clearly articulated if at all possible. Does the failure to reach a given accuracy level or a stated required precision mean that the observations otherwise have no value? Ideally what is needed is a statement of the reduction in value of an observation as its quality declines, but in practice this is difficult to provide.

Many of the requirements of GOFC/GOLD arise from statements of needs derived before the organization was set up (see section 3). Several derive from the work of the Data and Information System of the International Geosphere Biosphere Program (e.g., Townshend 1992). Given that most of the requirements were articulated over 5 years ago it is advisable they are all systematically revisited in the near future. An additional complication is that different users may require similar sounding products, but with different specifications. This has already been recognized to a degree by the Land Cover Implementation Team which has identified a suite of products which are similar in most respects but with very different spatial resolutions. It is important also to examine the burgeoning plans of organizations such as TCO (Terrestrial Carbon Observations) to ensure that their requirements are accurately described by GOFC/GOLD’s plans.

A word of caution is needed to prevent the excessive proliferation of products. It is clear that different end-users will require products with different specifications. Those concerned primarily with observations for biophysical models may require very different inputs than those concerned with governmental decision-support systems. The many stake-holders for forest and land cover observations each with particular domain-specific needs could potentially result in an unwieldy number of different products. Ideally it would be worth the effort to define those products which can be directly used by a wide cross-section of users. At any rate, it is desirable to define a relatively small number of basic products which can then be transformed to specific products tuned particular user needs. For example an annual land cover product at 250m which can be used for national forest inventory, can also be transformed into products at coarser resolutions, to suit the needs of regional and global modelers who need products at much coarser resolutions. It will be important to ensure that well defined rules of aggregation are defined to ensure that such users obtain the products they need. A further example from land cover is the possibility of creating products defining physiognomic characteristics as continuous fields (DeFries et al. 1999). These potentially can then be broken out into any arbitrary classes by applying user-defined thresholds.

In assessing requirements there will always need for a practical concern with what is feasible in technological terms and in terms of resources. A key role of GOFC/GOLD is therefore to identify the capabilities of current observational assets in meeting requirements and in assessing what improvements may be needed to match the requirements more closely.
5.2 AVAILABILITY OF OBSERVATIONS

Assessing whether observations once collected are available can be a relatively complex issue. First there is the question of whether the observational assets are in place to provide the needed data stream. One of the continuing goals of GOFC/GOLD is to try and ensure the latter happens. However it is not until an operational agency takes responsibility that this can be considered certain. Even then unless redundancy is built into the system there may still be gaps in the record as occurred in the mid 1990s for the afternoon AVHRR.

There are a wide range of factors which inhibit free unfettered access to useful data products. Data policy especially as it relates to costs and copyright is one major issue. The liberalization of copyright and lower costs of Landsat Enhanced Thematic Mapper data compared with products from earlier Thematic Mappers have greatly increased their use throughout the world (Goward et al., 2001). While the pricing is drastically reduced to reflect marginal cost recovery, access to large number of individual scenes adds up to a very large, possibly prohibitive, allotment of resources by the end user. Similarly, many in situ data also have to be paid for and although not necessarily individually expensive may be costly for a comprehensive data set. The current US policy of making satellite data available for the incremental costs of reproduction or even free if electronic means are used for acquisition is a worthy goal for all those responsible for distributing environmental data, but attention also needs to be paid to creative mechanisms to provide large quantities in bulk, or to provide resources to directly support those laboratories or institutions which can provide large quantities to many users cost-free.

The format of data can also hinder its availability. Apart from ensuring that appropriate data standards are adhered to, including the provision of high quality meta-data, different user communities may have very different technological abilities in handling data. One recent example was the supply of MODIS products in HDF format using an integerized sinusoidal (ISIN) projection. Many users found both the format and projection sufficiently problematic to make them reluctant to move from the lower quality AVHRR products to the MODIS ones. Many data are now being made available in alternative formats and projections and software has been provided to facilitate conversion to different projections.

To ensure that appropriate products are available for all of GOFC/GOLD’s user communities, the organization should work with agencies to ensure that appropriate products are being made available. It should also attempt to capitalize on existing assets such as the Global Land Cover Facility1 and the Tropical Rain Forest Information System in ensuring that improved optimized data products are made available. Attention should also be paid to the burgeoning GRID technologies which with their increasingly powerful middleware tools bring the prospect of providing for the user integrated data systems based on distributed, heterogeneous data systems (Foster and Kesselman, 1998).

Whereas there remain significant challenges in the availability of remotely sensed data the difficulties associated with obtaining many in situ data are much greater, because of their distributed nature, data policy including privacy issues as well

1 www.GLCF.umiacs.umd.edu ; www.bsrsi.msu.edu/trfic
as costs. Some initial steps have been taken and GTOS is increasing awareness of the availability of in situ data through its TEMS data base (GTOS 2003), and through the US Global Change Master Directory (NASA 2003). As yet most of these efforts are at the directory or catalog level rather than at the data base level, i.e. one can find out about data, but obtaining them normally requires interactions with multiple providers. For observational data sets to be regarded as being “available”, GOFC needs to set a minimum standard that they are openly available to any user without prejudice.

5.3 ALGORITHMS/ASSIMILATION ESTABLISHED

Raw data sets are often of limited value to most users who require higher level products. This requires the use of algorithms to convert the data into geophysical variables. For an algorithm to be regarded as acceptable it should have been subject to independent scientific scrutiny usually through publication in the scientific literature. For many products there has been relatively little scrutiny of the algorithms themselves, and relatively limited subsequent validation is usually carried out. For some products there has been active consideration of the algorithms used in generating products, notably in algorithms for active fire detection, especially contextual ones and also in estimating proportional cover estimation (Giglio et al., 1999). But other products algorithms have received little consideration, as in the case of land cover change. Although the current emphasis on the quality of the end-product is appropriate, there are clearly likely to be benefits from a more organized consideration of the relative merits of different algorithms in particular in relation to their physical soundness.

One other issue deserving comment is that of data assimilation to serve the needs of modelers. Currently assimilation for terrestrial processes is in its infancy but comparisons with atmospheric and oceanographic domains suggest that it is likely to become much more important in the future and that establishing appropriate procedures for assimilation from the numerous ones available (Liang 2003) is likely to be a future issue for GOFC/GOLD as an organization.

5.4 RESEARCH OR PROTOTYPE PRODUCTS CREATED

Typically the next stage involves the creation of a research product or a prototype of the long-term systematic product. The Pathfinder experience mentioned above as a precursor to GOFC/GOLD showed the efficacy of the experience gained with producing prototype products and end-to-end acquisition, processing and delivery mechanism. Several prototypes have been created during the last few years that either meet the specifications of GOFC/GOLD or represent a substantial advance towards these specifications. Where specifications are not fully met then GOFC/GOLD should formally indicate the extent to which the specifications are not met. Typically these products with a preliminary evaluation are reported upon in the refereed literature. The substantial time taken between submission of papers and publication may result in subsequent versions of products being made available before papers first appear.

An increasingly contentious issue is how to decide on the relative merits of different products created that nominally meet requirements but which are derived either from different instruments or using different algorithms. This currently is a major issue with respect to the detection of active fires (Ahern et al., 2001).
Differences between products may arise from differences in the time of sensing due to considerable diurnal variability of fires. Other differences may arise due to instrument sensitivity or to differences in the performance of the algorithms. Users of these products need guidance concerning their different capabilities and merits.

One way in which comparisons can be carried out is through validation, as described in the next section, but as we will see validation itself has its own challenges. Formal inter-comparisons between different products have been carried out in a number of areas including land cover and NPP. IGBP carried out extensive inter-comparisons of different products (Cramer et al., 1999). Where complex models are used with different observational inputs and no universally accepted validation set then deciding on the relative merits of products and what leads to the observed differences can be difficult.

5.5 VALIDATION AND QUALITY ASSESSMENT PROCEDURES ESTABLISHED AND APPLIED

Accuracy assessment of products often called validation is widely recognized as an important component in the development of operational observations, especially those from satellites, but related procedures to establish the quality of in situ observations are also routinely carried out. Comparison of satellite products is being continued by the CEOS Cal/Val Working Group on Land Product Validation (Morisette et al., 2002). Validation has always been regarded as a key activity for GOFC and has formed the subject of a number of workshops, notably for active fires and burn-scars (http://gofc-fire.umd.edu/products/pdfs/events/Lisbon_7_01_ExecSum.pdf) and for LAI (http://modis.gsfc.nasa.gov/MODIS/LAND/VAL/CEOS_WGCV/lai_intercomp.html). Whereas there has been agreement on the approaches that should be adopted a routine set of procedures and implementation of a continuing validation program has yet to be established. Nor has there been a structured comparison of the relative merits of different products of the same variable. In part this relates to the substantial costs of such work but these costs are relatively small compared with the costs of space missions.

In many respects the term “validation” is an unfortunate one since it implies that data sets are capable of being categorized as true or untrue, whereas almost all data sets hover somewhere between these two extremes. Additionally there are the practical difficulties of carrying out a comprehensive validation so we understand the reliability of a product for all geographic locations and for all time periods. In this respect it would be advisable if GOFC considered adopting the sort of scheme proposed by the MODIS Land Team (table 2) which represents different levels of effort (Justice 2002).

Assessing the accuracy of global products is an expensive and time consuming task and there are distinct advantages in international cooperation in terms of cost sharing and increasing the pool of available expertise for data collection (Justice 2002). Responsibility for validation lies primarily with the data producer and there is a need for accuracy assessment to be included in the overall costs of the satellite mission.
Table 2. Levels of validation (Justice 2002).

<table>
<thead>
<tr>
<th>Stage 1 Validation</th>
<th>Stage 2 Validation</th>
<th>Stage 3 Validation</th>
</tr>
</thead>
<tbody>
<tr>
<td>product accuracy has been estimated using a small number of independent measurements obtained from selected locations and time periods. Validation assessed locally under a limited range of geographic conditions for a limited period of time.</td>
<td>product accuracy has been assessed over a widely distributed set of locations and time periods Validation assessed over a significant range of geographic conditions and for multiple time periods and seasons.</td>
<td>product accuracy has been assessed and the uncertainties in the product well established via independent measurements in a systematic and statistically robust way representing global conditions. Validation assessed over the full range of global conditions for all time periods.</td>
</tr>
</tbody>
</table>

Validation is not the ultimate way of assessing the usefulness of a given set of observations. What one really wishes to know is how much difference a product makes to the end-user. In this context “field-testing”, by incorporating new observational products into actual models or decision-making systems, may reveal the most relevant guidance of their quality.

5.6 PRODUCT ADOPTED BY OPERATIONAL AGENCY

As we have discussed previously, since the goal of GOFC is to ensure long term systematic observation this ultimately means that an operational agency has to take on responsibility for the task. Reliance on research organizations will not usually lead to sustained observations because their mission to support innovative scientific work. Many of the observations specified by GOFC/GOLD have been created in prototype form by recent missions of NASA, ESA and NASDA, but this does not ensure their long term continuity. It is therefore important that the observations which have been proven useful by the research agencies be transitioned to the operational domain and that processes are put in place to facilitate that transition (NRC 2003).

There are varying degrees to which we may consider a product being adopted by an operational agency. Specifically we need to consider whether a product is adopted which is better than nothing but does not meet the requirements of GOFC/GOLD or whether the product meets specifications. For example NDVIs were regularly produced from the near infrared and red bands of the AVHRR sensor and since they were derived from an operational meteorological satellite they can be regarded as being operational in a sense, but the quality of this measurement definitely fell below that specified by most users (Justice and Townshend, 1994).

One of the most encouraging recent developments in terms of remote sensing has been the specification of the VIIRS sensor as the successor to the AVHRR (Townshend and Justice, 2002). VIIRS will be the operational sensor on board the NPOESS platform. It has many of the advanced characteristics of MODIS and its observations will be capable of creating many of the latter’s products. To ensure continuity between the MODIS of Terra and Aqua and the VIIRS of NPOESS, NASA is launching the NPOESS Preparatory Project (NPP) with the VIIRS sensor on board. For moderate resolution products several of the products specified by GOFC are likely to be generated operationally. However the environmental data products currently
specified may not have the long term internal consistency needed to monitor trends over extended time periods (NRC 1999, 2000).

Agreement by agencies to collect observations operationally unfortunately does not guarantee that this will in practice be done. The experience of the Global Climate Observing System working through the international structures of the WMO in gaining agreements from nations to collect key observations is cautionary. Neither the GCOS Surface Network (GSN) for measuring temperature nor the GCOS Upper Air Network (GUAN) has as yet achieved satisfactory performance despite formal long-term commitments made by national governments (GCOS 2003).

5.7 ROUTINE QUALITY ASSURANCE AND VALIDATION

Quality assurance and continuing validation need to be a continuing effort if the long term internal consistency of products and their relation to defined standards is to be maintained. Ideally this also should be carried out by an operational agency. Analogies of the way in which this is might be conducted are shown in the way that the European Center for Medium-Range Weather Forecasting monitors the performance of the GCOS Upper Air Network. In addition periodic scientific assessment of the results is also necessary. Examples of the required approach can be found in a critical review of measurements for long-term climate monitoring (Karl 1996).

5.8 ARE FURTHER IMPROVEMENTS NEEDED TO MEET REQUIREMENTS?

Operational products rarely meet all requirements so there will be a need periodically to assess the extent to which observational products meet evolving requirements and the feasibility of improving them. In this sense the whole process is a continuing iterative one with few observational types reaching the specifications needed by all users.

6 Progress in Meeting Our Goals

As previously stated the principal goals for GOFC/GOLD were set out approximately 4 to 5 years ago. Figure 2 summarizes the extent to which these goals have been met using a simple coding scheme to indicate whether substantive progress has been made for each of the stages outlined in section 5.

It is clear that substantial advances have been made in recent years. Few if any of the boxes would have been colored when the planning documents were first created. In some cases there is already a commitment to long term operational measurement and product generation as a result of some properties being accepted as environmental data records for NPOESS (Townshend and Justice, 2002). However it needs to be recognized that although they will be generated regularly they may not have the long term internal consistency needed for detecting long-term change (NRC 2000).

The GOFC/GOLD effort has stimulated advances in both science and applications, hence the recognition of the important role of observations to support LCLUC research on two important fronts: research and natural resource management/policy. In its report on Grand Challenges in Environmental Science, the NRC (NRC 2001) has identified land use dynamics as one of the emerging key research domains as a result of significant advances in the community readiness through increased observational capacity. Similarly, the NRC report, Down the Earth (NRC
MEETING THE GOALS OF GOFC

2002) has pointed to new and increased opportunities for the use of geographic information to support sustainable development. Moreover, international agencies, in particular the UN Food and Agriculture Organization are beginning to develop cooperative observation-intensive approaches to their operational mission for collecting data on the state of the world’s forests through the Forest Resource Assessment.

Figure 2a,b,c,d. Progress in meeting the goals of GOFC/GOLD (Dark gray indicates substantive progress; light gray equals partial progress and white means no progress yet.

![Fine Resolution Cover Products Table]

Another key issue, beyond the scope of this discussion, is the steps needed to ensure that the various products are made available to users in product delivery systems suited to their requirements and their technological level. Simply because archives exist holding data does not ensure that users will be able to utilize the products. Among the many challenges are the provision of sufficient bandwidth for users, that products are available in easily readable formats, that the projections are compatible with commonly available software, that the data sets have been subsetted so as not overwhelm the modest computing facilities of many users. There may be additional obstructions to use caused by charging policies and copyright.
## Coarse resolution land cover products

<table>
<thead>
<tr>
<th>GOFC/GOLD Products</th>
<th>GOF Spec.</th>
<th>Require</th>
<th>Observ</th>
<th>Algorithm</th>
<th>Prototype</th>
<th>Assessment</th>
<th>Operational</th>
<th>QA &amp; Val</th>
<th>Iterate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land cover classification</td>
<td>CNES W/S App 4, p.31</td>
<td>Y</td>
<td>MODIS/ Vegetation/ AVHRR</td>
<td>Y</td>
<td>MODIS standard product; OLC 2000</td>
<td>P</td>
<td>Y (MIRS EDR)</td>
<td>N</td>
<td>N</td>
</tr>
<tr>
<td>Forest density (continuous fields)</td>
<td>CNES W/S App 4, p.33</td>
<td>Y</td>
<td>MODIS/ Vegetation/ AVHRR</td>
<td>Y</td>
<td>MODIS product</td>
<td>P</td>
<td>N</td>
<td>N</td>
<td>N</td>
</tr>
<tr>
<td>Land cover change (indicator)</td>
<td>CNES W/S App 4, p.34</td>
<td>Y</td>
<td>P (MODIS 250m)</td>
<td>Y</td>
<td>N (available regionally)</td>
<td>N</td>
<td>N</td>
<td>N</td>
<td>N</td>
</tr>
</tbody>
</table>

May 28th, 2003

Canadian Forest Service, Edmonton

## Fire Products

<table>
<thead>
<tr>
<th>GOFC/GOLD Products</th>
<th>GOF Spec.</th>
<th>Require</th>
<th>Observ</th>
<th>Algorithm</th>
<th>Prototype</th>
<th>Assessment</th>
<th>Operational</th>
<th>QA &amp; Val</th>
<th>Iterate</th>
</tr>
</thead>
<tbody>
<tr>
<td>Active fire detection - daily (polar)</td>
<td>CNES W/S App 4, p.35</td>
<td>Y</td>
<td>Y</td>
<td>Y</td>
<td>MODIS, AVHRR, CNES, AATSR, VIRS</td>
<td>P</td>
<td>Y</td>
<td>P</td>
<td>N</td>
</tr>
<tr>
<td>Active fire detection - diurnal cycle (geostationary-polar)</td>
<td>FIRE IT web site</td>
<td>Y</td>
<td>Y</td>
<td>Y</td>
<td>GOES, VIRS</td>
<td>P</td>
<td>Y</td>
<td>N</td>
<td>N</td>
</tr>
<tr>
<td>Burnt area</td>
<td>CNES W/S App 4, p.36</td>
<td>Y</td>
<td>Y</td>
<td>Y</td>
<td>Globevis, GBA 2000, MODIS Regional</td>
<td>P</td>
<td>N</td>
<td>N</td>
<td>N</td>
</tr>
<tr>
<td>Emission product suite</td>
<td>FIRE IT web site</td>
<td>Y</td>
<td>P</td>
<td>?</td>
<td>N (available regionally)</td>
<td>N</td>
<td>N</td>
<td>N</td>
<td>N</td>
</tr>
<tr>
<td>Fire danger rating</td>
<td>FIRE IT web site</td>
<td>Y</td>
<td>?</td>
<td>?</td>
<td>N (available regionally)</td>
<td>N</td>
<td>N</td>
<td>N</td>
<td>N</td>
</tr>
</tbody>
</table>

May 28th, 2003

Canadian Forest Service, Edmonton
7 Conclusions

GOFC/GOLD has made considerable progress in laying out its requirements to meet the needs of its user communities. Its principal goal is to ensure the long term consistency of key terrestrial products. Its focus so far has been on land cover characterization and a variety of fire products. Considerable improvements have been made in most of the various stages which need to be completed to achieve these goals but in reality there are few products where an operational commitment has been made. Considerable efforts are needed to ensure the availability of products to user communities. Continued and concerted effort will be needed by the science community, operational data users, space agencies and data providers to ensure that the objectives of GOFC/GOLD are met. The ultimate objective is to ensure operational integrated satellite and ground based observing systems are put in place to provide the data and information needed for an improved understanding of the earth system and effective management of its natural resources.

To achieve the goals of GOFC/GOLD the research community has several key responsibilities. This book highlights many of the most important of these:

1) Carrying out research to improve our understanding of the nature and drivers of changes in land cover. Results of such work in this volume include the mapping of desertification (Prince), woodland expansion into grasslands (Wessman et al.), tropical deforestation (Skole et al.), agricultural development in arid lands (Hole and Smith) urbanization both locally (Seto et al.) and globally (Elvidge et al.) and the multiple roles of fire (Csiszar et al.). Creating improved models of land use and cover
change is also of high importance if we are to understand the drivers and impacts of change (Brown et al.).

2) Scoping and setting up initial regional observing systems to ensure both the gathering and use of data. Several case studies in this volume highlight the importance of the scientific rigor needed to design such systems (McGuire et al.) and the challenges especially in developing countries (e.g. (Laporte et al.; Krankina et al.; Sader et al.; Seto et al.).

3) Assessing the impacts of land cover change on the environment. These include the global carbon cycle (Houghton et al.), the hydrological cycle (Mustard and Fisher), climate (Bonan et al.) and biodiversity (Hansen et al.) and the complex interactions between land cover/use and population (Rindfuss et al.).

4) Linking the physical changes we observe to the human populations and their activities is also essential if we are to understand past and future trajectories of change, (Rindfuss et al.; Turner II et al.).

Above all this volume demonstrates that improved environmental observation systems must be underpinned by sound scientific research and that they must be designed and implemented not simply as passive recorders of change but to enable users, whether in the scientific, policy or management communities, to address questions on the nature, cause, impacts of land cover change and the ways in which all of these change in the future.

Acknowledgement. The authors wish to acknowledge the role that NASA and in particular the LCLUC program have played in supporting GOFC/GOLD and its regional network activities. Grateful acknowledgment is also given to the Canadian Space Agency and Canadian Forest Service for their support.

8 References


Bonan, G. B., R. S. DeFries, M. T. Coe and D. S. Ojima Land Use and Climate. This volume.

Brown, D. G., R. Walker, S. Manson and K. Seto Modeling Land-Use and Land-Cover Change. This volume.


Hansen, A., R. DeFries and W. Turner “Land Use Change and Biodiversity: A Synthesis of Rates and Consequences during the Period of Satellite Imagery.” This volume.

Hole, F. and R. Smith Arid Land Agriculture in Northeastern Syria: Will this be a tragedy of the commons? This volume.


Laporte, N. T., T. S. Lin, J. Lemoigne, D. Divers and M. Honzak Towards an Operational Forest Monitoring System for Central Africa. This volume.


Prince, S. D. Mapping Desertification in Southern Africa. This volume.


Sader, S. A., R. R. Chowdhury, L. C. Schneider and B. L. Turner II Forest Change and Human Driving Forces in Central America. This volume.


Turner II, B. L., E. Moran and R. Rindfuss Integrated Land-Change Science and Its Relevance to the Human Sciences. This volume.

Section II Observations of LCLUC: Case Studies
SECTION II: INTRODUCTION

OBSERVATIONS OF LCLUC IN REGIONAL CASE STUDIES

DAVID L. SKOLE, MARK A. COCHRANE

This section presents a series of regional case studies using remote sensing measurements and integrative analyses across a range of land use and cover change situations around the world. The knowledge derived from these cases is invaluable for building regional models of LCLUC and as a foundation for calibration and validation of global LCLUC remote-sensing products. The papers here frame key driving forces by linking observations to processes. Such regionalized case studies are critically important in refining our understanding of the details of LCLUC and, ultimately, for linking research to applications, problem solving and policy.

Regional case studies are important ways to develop the spatial and temporal detail necessary to model land use and cover change as an agent of global change. Unlike climate studies which have typically been global in scale, understanding the dynamics of land use and cover change requires finer resolution in time and space. The development of dynamic models of land use and cover change involves a much closer examination of fine scale processes, and the relationship between use and cover. Moreover, land use and cover dynamics have important spatial attributes that are only revealed at finer scales.

Most previous global efforts to model land cover change have been spatially and temporally aggregate in nature (c.f. 0.5° - 4.0°; decadal time steps). Research has focused on biogeochemistry (e.g. carbon flux), changes in ecosystem metabolism (e.g. NPP) or changes in water and energy balance (e.g. albedo or latent and sensible heat flux). Global ecosystem models have typically introduced land cover change as a simple forcing function derivative of the total area of land cover change. For example, most models use aggregated deforestation rates to drive carbon flux or changes in continental NPP. When a spatial context has been introduced into models, it has generally been done to treat the large-area variations in biomass classes found on land cover maps or for connectivity to basin-scale hydrology. These models have been invaluable for ascertaining the global-scale significance of land use and land cover change for forcing fluxes of radiatively important gases, climate forcing from land surface changes, or biome-wide ecosystem response to climate change. Arguably, the scale selected has been consistent with the questions they pose. But this generalized picture of global change, with respect to land use and cover change, is far from complete.

Models have not explicitly treated the spatially varying landscape mosaic left behind by a range of anthropogenic and natural disturbances acting simultaneously and interactively within a locale. As a consequence, much of the research and modeling is undertaken at a scale that overlooks the actual processes associated with land use and land cover change. To gain insight into the land use-land cover relationship several fine scale factors need to be included, such as: interaction between different types of land uses, spatial pattern and landscape patch relationships, the role of social factors at household to community levels, inter-annual variation in rates of change, and intra-annual variations in biophysical characteristics of cover. As the papers in this section
demonstrate, through focused regional case studies, it is possible to make accurate measurements of the magnitude and spatial pattern of land cover change, and also elucidate the important processes that are involved.

The collection of papers prepared for this section also provides an excellent cross section of approaches and methods for linking process level LCLUC dynamics with remotely sensed observations. The notion that one can meld these two types of observational data has captivated the research community for some time now, and it is apparent that groundbreaking work is now underway to actually develop and use what has only been discussed elsewhere. One of the more important framing concepts behind all of the papers is that which Turner and colleagues have termed socializing the pixel (Geoghegan et al., 1998). This conceptualization has been one of the more important methodological issues in current LCLUC research, and it has guided the development of a variety of specific approaches, many of which are illustrated in the papers for this section. In another way, this chapter lays out a pattern-to-process framework for LCLUC research, heavily grounded on a foundation of direct observations across many scales and coupled to detailed understanding of the processes which drive land use to result in the observed land cover changes.

Whether under a rubric of socializing-the-pixel or pattern-to-process, the work demonstrated in these chapters is leading to enhanced social science methods through improved spatial descriptions of landscapes and, in turn, enhanced biophysical observations of landscape change derived through an improved understanding of the human drivers. In this way, the spatially-mapped landscape becomes an important attribute, which defines the biophysical context to which the social context can be related. The comparison of the frontier case studies (Sader et al.; Laporte et al.; Krankina et al.; and McGuire et al.) and non frontier case studies (Seto et al.; Samek et al.) is an excellent portrayal of the differential relationships between the social and physical dimensions, and between use and cover relationships. In the former cases, socio-economic factors of use drive cover outcomes directly – for example with deforestation processes. In the latter cases, long histories of rice and other cultivation have resulted in tightly coupled physical-social landscapes, where the spatial configuration of human uses of land and the spatial configuration of the physical land cover are interrelated. The cases from the extensively managed grasslands in Africa and the Middle East and the intensively managed systems in the United States (Prince, Hole and Smith, and Wessman et al.), on another hand, portray the dynamics of changes in use, as well as intensification and management. For these regions, land cover change is not as dramatic as in forested ecosystems but the demonstrated changes in ecosystem composition, structure and potential utility of these lands are profound. The analyses of Skole et al. identify the close linkages between various components of the overall disturbance regime in Amazonia, with measurements being made of both direct conversion and more subtle degradation. The degree to which these emerge from different agents and also interact synergistically, for example the interrelationship between logging, deforestation and fire, provides important insights for modeling of land cover and land use change.

These studies show important differences in the dynamics of land cover and land use change (LCLUC) from place to place, without appearing to be idiosyncratic. Frontier areas are characterized by inter-annual changes in cover driven by land use change. Older regions, where the frontier gives way to more anthropogenically-
dominated landscapes, landscape dynamics are dominated by inter-annual variation in land uses, and intra-annual changes in cover characteristics. The notion of a tightly coupled human-natural landscape emerges, and casts aside the view of the unidirectional social “driver” causing the physical “impact”, which seems to prevail in so much of the current global change literature.

One of the key contributions made by direct observations of the Earth is an improved understanding of pattern and the spatial context of LCLUC. More than simply providing a basis for disaggregation, and hence improved quantification and measurement (indeed an important factor in its own right, when much of the basic measurements need to be made), spatially-explicit analyses provide vastly improved understanding of how landscapes evolve over time by incorporating the spatial interrelations between types of land uses, the agents of land use change, and the effects of interrelated disturbance on land cover. It may be argued that both the social sciences and the physical sciences have not emphasized the important interaction between landscape pattern and social systems over time, and hence, the relationship of past and current patterns of LCLUC to future trajectories. Remote sensing clearly provides a useful dataset in this regard. The coupling of remote sensing observations to socio-economic analysis of LCLUC is a way to capture the physical landscape and the patterns of transformation in space and time.

Lastly, one of the more important and under-examined issues is the relative roles of natural vs. social partitioning of the landscape. By placing human factors within the study of the global environment, we may yet come to know how the interactions between social institutions, which partition the landscape through title and ownership, and nature, which does so through topography, biogeochemistry and climate, play out across the landscape and over time. Analyses or models, which lack such a fundamental use-to-cover component, are incomplete and, due to their missing processes, incapable of providing credible forecasting and prediction of LCLUC. Regionalized case study assessments, of the kind presented in the following chapters, start to address such deficiencies and bring us ever closer to a true understanding of our evolving planet.
CHAPTER 4

FOREST CHANGE AND HUMAN DRIVING FORCES IN CENTRAL AMERICA

STEVEN A. SADER\textsuperscript{1}, RINKU ROY CHOWDHURY\textsuperscript{2}, LAURA C. SCHNEIDER\textsuperscript{2}, B.L. TURNER II\textsuperscript{2}

\textsuperscript{1}Department of Forest Management, University of Maine, Orono, ME 04469-5755
\textsuperscript{2}Graduate School of Geography and Marsh Institute, Clark University, Worcester, MA 01610 USA

1 Introduction

Three research projects, supported by the NASA Land Cover and Land Use Change (LCLUC) Science Program have been conducted in the Central America and Mexico (Mesoamerica) region. The first two projects were conducted in Northern Guatemala (Maya Biosphere Reserve) and the Southern Yucatan Peninsula Region (SYPR) of Mexico. The third project, focused on the Mesoamerican Biological Corridor, had broader coverage over the entire Central American region under a cooperative research memorandum of understanding between NASA and the environmental ministers of the seven Central American countries. This chapter will begin with an overview of the Mesoamerican Biological Corridor concept and the status of forest protection and forest changes using data analyzed for sample sites distributed throughout the region. Given the importance of protected areas in the region and the threats facing them, the bulk of the chapter will address the Northern Guatemala and SYPR sites located in and around two major Biosphere Reserves. Both of these case studies utilized time-series Landsat imagery combined with socio-economic household surveys and landscape level analysis to examine human driving forces that influenced forest and land cover/use changes. This research demonstrates that medium spatial resolution satellite imagery is well suited for monitoring tropical forests and remote biological reserves (Sader et al., 2001a; Turner et al., 2001).

2 Significance of the Mesoamerican Region

The Mesoamerican region covers approximately 768,990 square kilometers, stretching from the southern Yucatán Peninsula of Mexico through Panama (Figure 1). The Isthmus of Central America is comprised of more than 20 distinct ecological life zones (Holdridge et al., 1971) ranging from coral reefs, coastal wetlands, and Atlantic lowland wet forests to pine savannas, Pacific dry forests, and montane cloud forests. This land bridge connecting the North and South American continents contains only one-half of a percent of the world’s total land surface, yet its geographic position and variety of ecosystems combine to harbor about 7 percent of the planet’s biological diversity (Miller et al., 2001). The region’s globally significant biodiversity, combined with its small area and high degree of threat in terms of habitat loss, has prompted the conservation community to designate the region as a biological, as well as a
deforestation “Hot Spot”, to be prioritized for conservation planning and management (Achard et al., 2002; Wilson, 2002).

The region’s rapid population growth, human migration, and slash-and-burn agriculture have had detrimental effects on what forest remains. The 1980-1990 annual rates of deforestation in Central America and Mexico were among the highest in the world (FAO, 1993) and high rates of deforestation continued into the 1990’s (FAO, 1999; Sader et al., 1997, 2001a and 2001b; Sanchez et al., 1999; Roy Chowdury and Schneider, 2003). Forest fragmentation and loss of critical natural habitats (Sanchez et al., 1999; Sader et al., 2001b) present some urgency for the development of conservation management strategies, perhaps best achieved through a cooperative research agenda and scientific database for continuous monitoring of the Central American region (Miller et al., 2001).

2.1 MESOAMERICAN BIOLOGICAL CORRIDOR: A VISION OF SUSTAINABLE DEVELOPMENT AND HABITAT PROTECTION

Reserves and protected areas form the foundation for conservation strategy across the tropics. The effectiveness of these reserves in conserving biodiversity and landscape integrity relies on their proper design and management, as well as their protection and legal enforcement (Hansen and Rotella, 2002). Often, designated protected areas suffer from lack of connectivity with other vital habitats and protected reserves, lack of protection enforcement, land degradation in surrounding areas, as well as an incomplete
understanding of the drivers of that degradation (Sader et al., 1997; Liu et al., 2001; Hansen and Rotella, 2002). Large forest patches and core areas protect some of the most vulnerable plant and animal species that cannot survive in smaller patches and fragmented forest landscapes. Forest edges are more vulnerable to fire and effects of predators on nests or habitats of interior-dwelling species (Hunter, 1996; Noss and Csuti, 1997). Loss of connectivity between forest patches can limit migration of species. Monitoring of protected areas and characterizing the human and ecologically controlled processes in which they exist are crucial in conservation planning. (Kristensen et al., 1997; Bruner et al., 2001).

The origin of the Mesoamerican Biological Corridor (MBC) concept dates back to the 1980s, when the Wildlife Conservation Society (WCS) and Dr. Archie Carr III of the Caribbean Conservation Corporation joined forces to investigate the possibility of establishing a natural corridor to link protected areas throughout the range of the Florida Panther (Puma concolor coryi). The U.S. Agency for International Development (USAID) provided funds under the project known as Paseo Pantera to strengthen the management of protected areas throughout Central America (Carr 1992).

The MBC was officially established in 1997 by the presidents of the seven member countries as a crucial environmental region with a vision of integrating conservation, sustainable use, and biodiversity within the framework of sustainable economic development. The Central American Commission on the Environment and Development (CCAD), including the environmental ministers of Belize, Costa Rica, El Salvador, Guatemala, Honduras, Nicaragua and Panama, was established in 1989 and is responsible for coordinating regional environmental activities in Central America (CCAD, 1989). The CCAD and the region’s governments have endorsed the MBC concept and there are major donor-funded activities (e.g. World Bank, Global Environment Facility, United Nations Development Program, and USAID, among others) with various goals and commitments to the Corridor concept.

2.2 REGIONAL ANALYSIS OF FOREST COVER AND CHANGE IN MESOAMERICA

As mentioned earlier, an important function of the MBC is maintaining the connectivity of natural habitats throughout the region (Carr, 1992). Currently, the MBC is a network of braided natural habitats including hundreds of strips, many only 2 or 3 km wide, connecting large parks and reserves and small protected areas (Miller et al., 2001). What is the status of forest cover in the MBC? Are the national parks and biological reserves being threatened by deforestation? In an attempt to answer these questions, one of the early goals of the NASA-LCLUC Mesoamerican Biological Corridor project was to determine how much forest was contained within the MBC and how much forest clearing had occurred over the past decade (late 1980s to late 1990s).

2.2.1 Methods

A vector layer, acquired from the Environment and Natural Resources Ministry of El Salvador, containing polygons representing existing and proposed protected areas within the seven Central American nations provided the units of analysis for investigating changes in forest cover within the MBC. The analysis was based on sample sites across the region, represented by Landsat Worldwide Reference System (WRS) Scenes. The Landsat Thematic Mapper (TM) scene locations encompassed
approximately 25 percent of the Central America region and 33 percent of the all of the MBC administrative units in the region. The spatial distribution of the parks, reserves and other designated corridor units in the MBC, along with the WRS locations of the Landsat TM study sites, are presented in Figure 2.

Figure 2. Regional study sites for mapping and change detection efforts in Central America. The study sites correspond to a sample of Landsat WRS scenes across the region, and are shown in relation to the protected and proposed units of the Mesoamerican Biological Corridor (adapted from Sader et. al., 2001b).

For each site, land cover was classified using both Level I (4 classes: forest, shrub, non-forest, and water) and nested Level II (ranging from 15 to 25 classes) classification schemes. Land cover classification techniques made use of a combination of unsupervised clustering, supervised training using reference data from the World Bank Ecosystems Map of Central America, cluster busting, and post-classification editing. All classifications were based on Landsat-5 TM data. The change detection results were generated by a simple post-classification comparison of two dates (late 1980’s to late
2.2.2 Forest cover and change in the 1990s

Among the TM study sites (Figure 2), the forest cover ranged from 13.6% (Path 19/Row 50) to 82.6% (Path 16/Row 50) of the total land area classified. In the mid to late 1990s, the average forest cover over all study areas was 57.3% (1996-98). The time periods for the change detection analysis varied from 7 to 11 years, with the median period being 10 years. Annual forest clearing rates ranged from 0.16% (site 16 / 50) to 1.28% (Path 19/Row 50). The mean clearing rate for the TM study sites combined was 0.58% per year. Annual rates of clearing were calculated by the following formula:

\[
\frac{C_t}{F_t} \times 100
\]

where \(C_t\) is the area cleared during time period \(t\); \(F_t\) is the area of forest remaining at the beginning of time period \(t\), and \(Y_t\) is the number of years in time period \(t\).

For reporting purposes, the original administrative units of the MBC were grouped into five zones (Table 1). The total area of forest cover, the percentage of forest cover, and annual forest clearing rates were calculated for each MBC zone. The forest cover ranged from 57.2% in undeclared protected areas to 93.8% in current parks and reserves (mean = 80.4%). For comparison, only 30.8% of the total land area classified outside the MBC zones was under forest cover. Among the five MBC zones, annual forest clearing rates were lowest in designated extractive reserves (0.15%) and clearing was highest in areas proposed for addition to the protected corridor (0.57%). New protected areas, parks and reserves, and protected areas without declaration all had similar clearing rates of 0.22%, 0.25%, and 0.26% per year, respectively. Forest clearing outside of the MBC zones was substantially higher (1.4% per year) than the areas within the MBC (0.26% per year).

The higher percentages of forest cover remaining and the lower deforestation rates for parks and reserves may be an artifact, in part, of the conditions that led to the original designation of these areas as reserves. For example, they tend to be the most remote forests furthest from population centers, often occupying steeper slopes and soils less suitable for agricultural uses. However, the lowest deforestation rates in extractive reserves likely reflects both historical and present day reliance on non-timber forest products, practiced by local inhabitants, (for example, chicle, xate and allspice collection from the Maya Biosphere Reserve) as a principle source of income and an alternative to swidden cultivation that correlates with higher deforestation rates (Schwartz, 1990; 1998).

Although this study indicates that protection status affords some conservation of forest cover in the MBC (80% forest cover inside compared to 31% outside), there are geographic locations in the region and some zones that have very low proportions of forest cover. One finding of particular interest was the percentage of forest remaining and the forest clearing rates in the “proposed” corridor areas. This zone had only 59% forest cover (of the classified land area in the sample), along with an annual clearing rate that was more than twice as high (0.57%) as any others. If these proposed corridors are to be linked to the currently protected areas of the MBC then the forest cover losses
and fragmentation occurring in these zones should warrant thoughtful consideration by Central American governments and the conservation community.

This analysis did not specifically address forest fragmentation, however it is likely that fragmentation in the region has been increasing along with the documented forest clearing. Some evidence comes from Sanchez et al. (1999) who compared deforestation rates and the extent of fragmentation inside and outside protected areas in the Sarapiqui region of Costa Rica. Their study indicated higher fragmentation outside of the protected zones with an increase in the number of patches and a decrease in mean size from 0.95 to 0.25km$^2$.

Table 1. Annual forest clearing rates for protection zones of the proposed Mesoamerican Biological Corridor; all Landsat study sites combined, based on a 10 year median (from Sader et al., 2001b).

<table>
<thead>
<tr>
<th>Protection Zone Status</th>
<th>% Forest Remaining</th>
<th>%/Year Forest Clearing</th>
</tr>
</thead>
<tbody>
<tr>
<td>Parks and Reserves</td>
<td>93.77%</td>
<td>0.25%</td>
</tr>
<tr>
<td>Undeclared Protected Areas</td>
<td>57.23%</td>
<td>0.26%</td>
</tr>
<tr>
<td>Extractive Reserves</td>
<td>91.48%</td>
<td>0.15%</td>
</tr>
<tr>
<td>New Protected Areas</td>
<td>76.17%</td>
<td>0.22%</td>
</tr>
<tr>
<td>Proposed Corridor</td>
<td>58.87%</td>
<td>0.57%</td>
</tr>
<tr>
<td>MBC Total</td>
<td>80.42%</td>
<td>0.26%</td>
</tr>
<tr>
<td>Outside the MBC</td>
<td>30.79%</td>
<td>1.44%</td>
</tr>
<tr>
<td>7 Countries Total</td>
<td>57.27%</td>
<td>0.58%</td>
</tr>
</tbody>
</table>

3 Southern Yucatan Peninsula, Mexico and Northern Peten, Guatemala: Case Studies in Land Cover and Land Use Change

The forest cover and change rates presented in the Mesoamerica regional analysis tell us about “where” and “when” the forest conditions occurred but very little about “why” or “how” humans were influenced to make decisions about LCLUC on the landscape. To understand LCLUC dynamics operating in the region, two NASA funded projects were conducted in adjacent study areas; one in the Southern Yucatan Peninsula of Mexico and the other across the border into the northern Peten district of Guatemala (Figure 1). It is not our intention to provide complete coverage of either project. Instead we focus on forest change rates and trends in combination with socio-economic data, landscape level analysis and modeling techniques to examine how humans are influenced to make decisions to clear forests primarily for subsistence agriculture land use in the region. The reader is referred to the literature cited in this section for more specific information on methods and detailed results that are not presented here.
3.1 PHYSICAL DESCRIPTION AND LAND USE HISTORY OF LA SELVA MAYA

The adjacent forests in southern Mexico, northern Guatemala, and Belize, an area known as La Selva Maya constitutes the largest contiguous tropical moist forest remaining in Central America (Nations et al., 1998). Along with supporting a rich diversity of plant and animal species, the Maya Forest contains some of the world’s most significant archeological sites for one of the most developed civilizations of the time, the Classic Maya period of A.D. 250 to 900. A thousand years ago, a postulated combination of factors including population growth, political instability and warfare, environmental destruction, and climate change may have led to the collapse of the ancient Mayan civilization (Rice 1991; Culbert 1993). A similar combination of factors threatens the forest and its current inhabitants today, despite a much lower population and a much shorter time frame (Sever 1998).

The region is a karst landscape dominated by rolling uplands, clay-filled depressions and subsurface drainage (Foster and Turner, 2003). The Sierra del Lacandon is the only significant mountain range along the northwest border of Guatemala with Chiapas, Mexico. Dominant natural vegetation types include tropical broadleaf evergreen forest, semi-deciduous broadleaf forest and scrub-shrub broadleaf vegetation, the latter two occurring in seasonally flooded swamps (Lundell 1937). Two major types of mature forests: upland medium to high-statured on well-drained soils and; lowland short-statured or bajo forest on seasonally flooded soils dominate the region, and are characterized more by differences in species abundance and forest structure than species diversity (Pérez Salicrup and Foster, 2000; Perez Salicrup, 2003). Trees in medium to high forest grow taller than those in the bajo, where seasonal inundation limits growth and produces a shorter stature. The two forest types share approximately 80% of tree species, but different relative abundances of each tree species. Overlap between the two forest types, however, indicates that they occupy a continuum along an environmental gradient, apparently determined by local topography, soils and soil moisture (Schulze and Whitacre, 1999). A marked north-south precipitation gradient, the karst geology, as well as natural (hurricanes, fire) and human (swidden cultivation, fire) disturbance regimes together influence the region’s vegetation and nutrient cycling.

Although natural disturbance caused by hurricanes may be important throughout the Yucatán peninsula (including northern Guatemala), on a longer term perspective, the role of human disturbance regimes outweighs the relatively low frequency and intensity of wind disturbance (Foster et al., 1997; Whigham et al., 1998; Foster et al., 1999; Boose et al., 2002). Forest composition in the early part of the 20th century reflected traces of Mayan occupation from 1000 BC to 600-900 AD (Lundell, 1934) including abundant economic species presumably important in Maya orchard-gardens (Gómez-Pompa et al., 1987; Whitmore and Turner, 2002) and common stands of species, such as ramón (Brosimum alicastrum), characteristic of Maya disturbed soils (Lambert and Arnason, 1981). Perhaps the most striking legacy of early 20th century timber extraction activities in these forests is the drastically reduced population of mahogany (Swietenia macrophylla) and Spanish cedar (Cedrella odorata) (Snook, 1998), species documented to be locally abundant in upland forest before the peak of the timber operations (Miranda 1958; Pennington and Sarukhan, 1968).
3.2 THE MAYA BIOSPHERE RESERVE, PETEN, GUATEMALA

El Petén, Guatemala’s largest and northernmost department, covers 36,000 square kilometers of mostly lowland tropical forests and wetlands. Today, the Petén supports a low but rapidly growing population relative to the more crowded and heavily deforested southern highlands of Guatemala. The traditional life of these people has included mainly shifting cultivation agriculture and the harvest of non-timber forest products, such as chicle, xate, and allspice. This “forest society” of the Petén (Schwartz 1990), and the livelihood of its people, is inextricably linked to the fate of the forest. The forest is continually being cleared and its resources greatly taxed as human migration and the expansion of the agricultural frontier threatens the people and environment of the northern Petén (Sader et al., 1997).

In the face of this population expansion and destruction of forests, the Maya Biosphere Reserve (MBR) was established by congressional decree of the Guatemalan government in 1990 to preserve the remaining intact forest and the rich biological and cultural resources that it holds. Spanning approximately two million hectares, the MBR is a complex of designated management units including five national parks, four biological reserves, a multiple use zone, and a buffer zone (Figure 1). Within the multiple use zone and buffer zone, areas were further delineated into forest concessions where management responsibilities were delegated to the communities there-in. Swidden cultivation, cattle grazing, and some timber extraction is practiced within most concessions. Some older communities have developed less dependence on swidden cultivation and cattle grazing and include more agroforestry and harvesting of non-timber forest products (Schwartz, 1990). Although there is no ejido system of land tenure as in the bordering Mexican states (see section below on the Mexican study site), the concessions have a similar structure in use of agricultural plots allocated to families.

Sader et al. (1997 and 2001a) and Hayes et al. (2002) have monitored over ten years of deforestation in the MBR via time series Landsat image analysis. In comparison to 1980’s deforestation estimates (FAO 1993) for Guatemala (1.7%/year), Sader et al. (2001a) reported that the annual rate of forest loss within the MBR was lower, however, annual forest clearing estimates for the buffer zone were well above the averages reported by the FAO for Guatemala. The rates and trends of forest change were observed by comparing the four change detection time periods and reported for all management units in the MBR (Table 2). The annual forest clearing rates for the MBR (excluding the buffer zone) increased from 0.04% in 1986-90 to 0.23% in 1990-93 then remained nearly constant at 0.33% in 1993-95 and 0.36% per year in 1995-97. Forest clearing in the multiple use zone in the last two time periods, was stable at 0.25% per year. The clearing rate in the buffer zone increased from 0.74% to 2.71%, 3.28% and 3.78% per year in the four sequential time periods. The accuracy of the change detection results exceeded 85 percent (Sader et al., 2001a).

3.3 CALAKMUL BIOSPHERE RESERVE, CAMPECHE AND QUINTANA ROO, MEXICO

The LCLUC-SYPR project focuses on a 10,000 km² swath across the base of the Yucatán peninsula, capturing significant portions of the two Mexican states of Campeche and Quintana Roo that are now incurring major forest conversion (Figure 3). Except for seasonal extraction of chicle, the study area remained ephemeraly used
from the time of ancient Maya abandonment until the middle of the 20th century, when low-scale logging of tropical hardwoods, especially mahogany and Spanish cedar began. Major population growth and land-use change did not begin until the end of 1967 when a paved highway connecting the two coasts was built through the region, and land grants established newer ejido settlements (the ejido is a communal system of land tenure recognized at the federal level). This opened up the forest lands to farmers, private ranchers, and ultimately, large-scale development projects in the 1980s. Major deforestation ensued, including the clear-cutting of large wetland (bajo) forests. The failure of these projects as well as concern about the scale of deforestation led to the creation, in 1989, of the Calakmul Biosphere Reserve (figure 1). Ejidos in the region of the Calakmul Biosphere Reserve increasingly reflect a complex set of strategies that combine agricultural crops with fruit production and economically viable timber and nontimber forest products in experimental agroforestry systems. However, the land use change that continues to dominate the area is agricultural deforestation and to some extent pasture.

Table 2. The Maya Biosphere Reserve Management Units: Area and Annual Forest Clearing Rates (% per year) from 1986 to 1997.

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Tikal National Park (N.P.)</td>
<td>55,005</td>
<td>2.62</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Laguna del Tigre N.P.</td>
<td>289,912</td>
<td>13.94</td>
<td>0.01</td>
<td>0.28</td>
<td>0.57</td>
<td></td>
</tr>
<tr>
<td>El Mirador N.P.</td>
<td>55,148</td>
<td>2.63</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Sierra del Lacandon N.P.</td>
<td>191,867</td>
<td>9.57</td>
<td>0.13</td>
<td>1.18</td>
<td>1.26</td>
<td>0.78</td>
</tr>
<tr>
<td>Rio Azul N.P.</td>
<td>61,763</td>
<td>2.95</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Cerro Cahui Reserve</td>
<td>555</td>
<td>0.03</td>
<td>0.20</td>
<td>0.06</td>
<td>0.00</td>
<td>0.26</td>
</tr>
<tr>
<td>El Zotz Reserve</td>
<td>34,934</td>
<td>1.69</td>
<td>0.04</td>
<td>0.05</td>
<td>0.03</td>
<td>0.18</td>
</tr>
<tr>
<td>Dos Lagunas Reserve</td>
<td>30,719</td>
<td>1.47</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Laguna del Tigre Reserve</td>
<td>45,168</td>
<td>2.17</td>
<td>0.00</td>
<td>0.00</td>
<td>0.01</td>
<td>0.08</td>
</tr>
<tr>
<td>Multiple Use Zone</td>
<td>826,351</td>
<td>40.08</td>
<td>0.05</td>
<td>0.16</td>
<td>0.25</td>
<td>0.25</td>
</tr>
<tr>
<td>Buffer Zone</td>
<td>408,973</td>
<td>22.85</td>
<td>0.74</td>
<td>2.71</td>
<td>3.76</td>
<td>3.28</td>
</tr>
<tr>
<td>Total Area</td>
<td>2,000,335</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Regional land-cover maps were created for the SYPR to derive estimates of the major land changes using classified Landsat TM scenes. Two mosaics of the region were created using the TM scenes: one for the mid-1980s using scenes from 1984, 1985, 1987 and 1988, and the other for the mid-1990s using scene dates 1994, 1995, 1996 and 1997. The combination of scenes from different dates provides a more complete picture with less cloud coverage and solves some seasonal problems in class
separability, such as those emerging from the deciduous behavior of both bajo and medium forest. Land-cover changes from 1987 to 1997 were estimated for 6 classes: bajo forest, upland forest, mid-late succession (4-15 yrs), one savanna class, agriculture (cropland-pasture-early successional regrowth), and bracken fern. However, only regional forest change is reported here.

Results of the time-series Landsat observations for the SYPR indicated that 6.1% of the forest was cleared in the decade from 1987 to 1997, reflecting the increased agricultural activity in the southeastern section of the study area (Figure 3). When successional regrowth is included in the derivation of net deforestation rates, a 2.9% rate of deforestation results for the 10-year period. Annual rates of deforestation derive to 0.29%, which is nearly the same as the 1987-97 annual estimates reported earlier for the MBR (excluding the buffer zone). There is considerable spatial variation in the average deforestation patterns described above (Figure 4). For example, rates of forest clearing were significantly higher in the southeastern ejidos, along the eastern edge of the Calakmul Biosphere Reserve. In this area 15.8% of extant forest was removed between 1987 and 1997. Adjusting for secondary succession, net deforestation for this area lowers to 11.1%, more than three times higher, nevertheless, than the deforestation rate for the total study area. Given the relatively short period of observation (10 years), the status of secondary succession, whether transitory or persistent, cannot be directly determined.

While large government projects were important in deforesting wetland forests before the mid-1980s (Turner et al., 2001), subsequent human disturbance has focused almost solely on upland forests. Over the last decade examined, the amount of
cultivated lands taken from mature upland forests appears to have decreased, and the focus of cultivation shifted to successional growth, especially along the southern roadway. This shift may suggest a reduction in the swidden fallow cycle, also indicated by a four-fold increase in area invaded by bracken fern and more intensive chili-swidden cultivation.

Figure 4. SYPR regional and household econometric models of deforestation (adapted from Geoghegan et al., 2003).

Forest succession may be impeded by the increasing intensity of land clearing and the proliferation of successful invaders such as bracken fern (*Pteridium aquilinum*) Kuhn and *Viguiera dentata*. Resistant to fire and disease and tolerant of a broad range of climates and soils, these species become dominant under slash and burn agriculture (Niering and Goodwin, 1974; Putz and Canham, 1992; Lugo, 1997; Perez-Salicrup, 2003). In parts of the SYPR, the invasives inhibit forest regeneration, cause farmers to abandon the affected patch, and often promote the cutting of new forest to compensate for the cultivated area lost. How long these parcels remains “lost” for cultivation and/or return to forest is not known.

### 3.4. SYPR REGIONAL AND HOUSEHOLD MODELS

This section reviews significant results of models that attempt to explain human behavior or biophysical factors that influence forest clearing. Models of land use change often begin with explanatory models of human behavior that address where, when, and why land use-cover change occurs (Irwin and Geoghegan, 2001). The SYPR
project goals were to explain and predict land use change through two distinct econometric modeling approaches: (1) use individual household surveys at the household scale of analysis to understand and model land managers’ decisions, linking their outcomes directly to TM imagery via a geographic information system (GIS), and (2) to model at the regional scale using available remote sensing and census data (Geoghegan et al., 2001; Geoghegan et al., 2003). Underlying are economic theories built on principles of a maximizing land manager, either profit maximization if the land manager is engaged in the market, or utility maximization in the case of thin markets and subsistence cultivation (for a description of SYPR households and market cultivation, see Vance et al., 2003 and Keys, 2003). In both models, the dependent variable is the pixel transitioning from forest to non-forest as derived from TM imagery (Roy Chowdhury and Schneider, 2003), and the independent variables are biophysical (soils, topography, rainfall, forest type context) and socio-economic factors (demographics, wealth and infrastructure, education, ethnicity, welfare, subsidies, tenure, amount and location of land).

The regional model extends over the entire study area and links the satellite imagery with biophysical data and socio-demographic data from government census in a discrete choice logit model of deforestation. The regional model uses as the dependent variable the results from GIS change analysis for two different time periods of TM images, and focuses exclusively on the location of land use change. On the other hand, the household model focuses on the parcels associated with the household survey data. This approach uses the same biophysical data of the regional model, but also uses the richer socio-demographic data derived from the linkage of individual farm plots and the satellite imagery. The household model is slightly more sophisticated than the regional one as it focuses on both the location and timing of land-use change (Irwin and Bockstael, 2001) by tracking the land use associated with each particular pixel through multiple TM images over time, in a household hazard modeling framework.

Results from the regional logit and household models are summarized in Tables 3a and b (Geoghegan et al., 2003). It is of interest to note how each similar explanatory variable increases or decreases the probability of deforestation in both of the model specifications. Demographic variables in both models indicate that male population increases the probability of deforestation in a statistically significant manner, while female population decreases the same probability. While the estimated coefficients on the female population variable differ between the models, the overall results are similar, an unsurprising result given that the majority of households in the region are semi-subsistent producers, for whom family members simultaneously represent a source of labor as well as demand for outputs from the agricultural plots.

The biophysical variables (Table 3a and b) are all statistically significant, and all but one has the same sign in both models. The primary forest variable is negative in both models, suggesting that farmers prefer to clear secondary forest. As expected, superior upland soils and greater average precipitation increase the probability of deforestation.

Higher elevation is estimated to lower the likelihood of deforestation in both models, as it tends to be related to more rugged terrain, shallow soils and less soil moisture, all factors inhibiting yields. The only biophysical variable that differs between the two models is pixel slope. The logit model yields the counterintuitive result that increased slope increases the probability of deforestation. Thus the household hazard model might yield the more consistent result that at the scale of an individual
parcel, slope decreases the probability of deforestation, but at the regional level, the more important factors of soil and forest type better explain the clearing decision.

Table 3a. Econometric Results of Deforestation as Explained by Socio-economic, and Biophysical Variables from Regional Logit Model (adapted from Geoghegan et al., 2003)

<table>
<thead>
<tr>
<th>Dependent Variable:</th>
<th>Estimated Coefficient</th>
<th>t - statistic</th>
</tr>
</thead>
<tbody>
<tr>
<td>Deforestation</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1. Socio-economic</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Forest Extension (dummy)</td>
<td>-.0795</td>
<td>-5.96</td>
</tr>
<tr>
<td>Male population (per ejido)</td>
<td>.0035</td>
<td>21.73</td>
</tr>
<tr>
<td>Female population (per ejido)</td>
<td>-.0004</td>
<td>-2.22</td>
</tr>
<tr>
<td>Spanish speaking (per ejido)</td>
<td>-.0041</td>
<td>-69.04</td>
</tr>
<tr>
<td>Forest Extension (dummy)</td>
<td>-.0098</td>
<td>-39.26</td>
</tr>
<tr>
<td>Ejido size (# pixels)</td>
<td>-2.01e-06</td>
<td>-88.26</td>
</tr>
<tr>
<td>Percent with water (hh/ejido)</td>
<td>-.0048</td>
<td>-33.82</td>
</tr>
<tr>
<td>Percent with electricity (hh/ejido)</td>
<td>-.0006</td>
<td>-4.49</td>
</tr>
<tr>
<td>Percent receiving credit (hh/ejido)</td>
<td>.0038</td>
<td>14.60</td>
</tr>
<tr>
<td>Good soil (dummy)</td>
<td>.3281</td>
<td>53.37</td>
</tr>
<tr>
<td>2. Distance</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Distance to roads (m)</td>
<td>-.0001</td>
<td>-109.03</td>
</tr>
<tr>
<td>Distance to nearest ag land (m)</td>
<td>-.0021</td>
<td>-201.97</td>
</tr>
<tr>
<td>Distance to market (m)</td>
<td>-4.45e-06</td>
<td>-12.72</td>
</tr>
<tr>
<td>3. Biophysical</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Primary forest (dummy)</td>
<td>-.8773</td>
<td>-150.74</td>
</tr>
<tr>
<td>Elevation (m amsl)</td>
<td>-.0059</td>
<td>-91.57</td>
</tr>
<tr>
<td>Precipitation (mm)</td>
<td>.0012</td>
<td>22.24</td>
</tr>
<tr>
<td>Slope (degrees)</td>
<td>.0145</td>
<td>13.15</td>
</tr>
<tr>
<td>Number ag pixels (5x5)</td>
<td>.0592</td>
<td>67.11</td>
</tr>
<tr>
<td>Constant</td>
<td>-2.099</td>
<td>-30.35</td>
</tr>
</tbody>
</table>

Pseudo $R^2$ 0.2264
Number of observations 3944875

3.5 FOREST CLEARING RELATIONSHIPS WITH BIOPHYSICAL VARIABLES IN THE MBR

Some of the same relationships of human preference to clear secondary forest and avoid wet areas (bajos) found in the SYPR study were also observed in the Maya Biosphere Reserve study site. In the MBR study, exploratory models were used to examine relations between a landscape variable (forest type) and biophysical variable (distance to road and river access) to forest clearing over time. The study area contained several community concessions near the center of the MBR encompassing an area of approximately 514,000 hectares.
Table 3b. Econometric Results of Deforestation as Explained by Socio-economic and Biophysical Variables from Individual Household Hazard Model (adapted from Geoghegan et al., 2003)

<table>
<thead>
<tr>
<th>Dependent Variable:</th>
<th>Estimated Coefficient</th>
<th>t - statistic</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>1. Socio-economic</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Average # of males &gt; 11 over interval</td>
<td>.0145</td>
<td>1.63</td>
</tr>
<tr>
<td>Average # of females &gt; 11 over interval</td>
<td>.0365</td>
<td>3.88</td>
</tr>
<tr>
<td>Average # of children &lt; 12 over interval</td>
<td>.0270</td>
<td>5.40</td>
</tr>
<tr>
<td>Duration of occupancy</td>
<td>-.0413</td>
<td>-13.20</td>
</tr>
<tr>
<td>Duration of occupancy squared</td>
<td>.0005</td>
<td>8.49</td>
</tr>
<tr>
<td>Plot size (# of pixels)</td>
<td>-.0002</td>
<td>-16.77</td>
</tr>
<tr>
<td>Percent of interval owning chain saw</td>
<td>.1366</td>
<td>4.55</td>
</tr>
<tr>
<td>Percent of interval owning vehicle</td>
<td>.4522</td>
<td>13.09</td>
</tr>
<tr>
<td>Education of household head</td>
<td>.0261</td>
<td>9.81</td>
</tr>
<tr>
<td># of household members w/&gt; 8 years education</td>
<td>.0322</td>
<td>5.40</td>
</tr>
<tr>
<td>Native Spanish speaker</td>
<td>-.1005</td>
<td>-3.93</td>
</tr>
<tr>
<td>Percent of interval receiving government credit</td>
<td>-.2248</td>
<td>-7.81</td>
</tr>
<tr>
<td><strong>2. Distance</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Distance household to plot</td>
<td>-.0452</td>
<td>-27.56</td>
</tr>
<tr>
<td>Distance ejido to nearest market</td>
<td>-.0723</td>
<td>-36.72</td>
</tr>
<tr>
<td><strong>3. Biophysical</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Primary forest</td>
<td>-.6861</td>
<td>-39.62</td>
</tr>
<tr>
<td>Upland soil</td>
<td>.2566</td>
<td>10.62</td>
</tr>
<tr>
<td>Elevation</td>
<td>-.0104</td>
<td>-21.92</td>
</tr>
<tr>
<td>Slope</td>
<td>-.0253</td>
<td>-6.33</td>
</tr>
<tr>
<td>Precipitation</td>
<td>.1453</td>
<td>19.99</td>
</tr>
<tr>
<td>Constant</td>
<td>-3.5278</td>
<td>-8.70</td>
</tr>
<tr>
<td>Likelihood Ratio Chi²(31)</td>
<td>18641.66</td>
<td></td>
</tr>
<tr>
<td>Number of observations</td>
<td>115017</td>
<td></td>
</tr>
</tbody>
</table>

Hayes et al., 2002 demonstrated that the area of *alto* (high) and *medio* (medium) forest cleared was higher than *guamil* (fallow fields or early second growth) and *bajo* forest. In examining the use (clearing) of the different forest types for establishing farm plots, the relationship of clearing to forest cover type proportion available is helpful in illustrating trends. For example, forest closer to access corridors was considered to be “more available” than forest further from access. Given that the proximity relationship of clearing to points of access is a relevant factor in the analysis of forest type clearing, Hayes et al., 2002 first tested the general pattern of forest clearing in proximity to access.

Total area cleared, all time periods combined (1986-90-93-95-97), was plotted against distance from access, in kilometers. A curvilinear relationship was apparent, with area cleared decreasing exponentially as distance increased. Regression analysis indicated that distance from access was significantly and negatively associated with area cleared (coefficient = -29.98), with an R-square of 0.734. Distance squared was also significant to area cleared, with a negative coefficient and an R-square of 0.539. The strongest relationship was obtained by using the log transform of area cleared.
against distance from access (km), resulting in a coefficient of –0.53 and an R-square of 0.951 (Hayes et al., 2002).

Figure 5. The relationship of the percent of each cover type cleared to the percent available at 1 pixel (0.03km) intervals from roads and rivers. Mean residuals of the model suggest the relative human preference or avoidance of clearing a cover type (adapted from Hayes et al., 2002).

Having established the relationships of forest clearing over time from distance to access, it is informative to consider these effects in exploring the question of how different cover types are utilized (cleared) for establishing farm plots. However, the area of each cover type available will change over time and the amount of each cover type cleared will vary at different distances from access. Therefore, Hayes et al., 2002 hypothesized that if farmers showed no preference for clearing a particular forest type over another in establishing plots, the area of each cover type should be equally proportional to its availability. The percent of available area cleared in each cover type was plotted against the percent of total available area represented by each cover type, at each 1 pixel interval from access. They expected the points to fall along a line representing a 1:1 relationship of percent cleared to percent available. Points deviating from this line were examined by their residuals. Proportional clearing greater than availability had positive residuals, while proportional clearing less than that available had negative residuals, and equal proportions were close to zero (Figure 5). The mean

<table>
<thead>
<tr>
<th></th>
<th>alto</th>
<th>medio</th>
<th>bajo</th>
<th>guamil</th>
</tr>
</thead>
<tbody>
<tr>
<td>mean residual:</td>
<td>5.36%</td>
<td>-0.51%</td>
<td>-9.74%</td>
<td>8.14%</td>
</tr>
<tr>
<td>t</td>
<td>12.224 *</td>
<td>-1.607</td>
<td>-43.977 *</td>
<td>22.724 *</td>
</tr>
</tbody>
</table>

* significant at the 0.05 level
residual of bajo forest clearing showed the greatest deviation from what was expected based on availability (-9.7%); the significantly negative residuals suggesting that bajo is avoided in farm plot establishment. Positive mean residuals and significant t-test results for alto forest and guamil types (5.4% and 8.1%) indicate that a greater proportional area was cleared in these types than their availability at each interval from access. This suggests a preference for these types in plot establishment, with a slightly higher propensity for using guamil-fallow fields where available.

Information from a household survey in the study area (Schwartz, 1998) provided the socio-economic context for the analysis of land cover and land use change from the time-series database. Analysis of the database provided a general examination of the use of clearing and fallow in adaptive strategies. For example, the overall trend toward avoidance of clearing in bajos suggests a lack of adaptation by migrants from the highlands of southern portions of Guatemala to traditional lowland farming techniques. Indeed, there is evidence that the Ancient Mayans farmed the seasonally flooded, lowland bajos (Turner 1983; Sever 1998), and that the techniques that they employed may be an important clue to the question of how the civilization supported much higher population densities than exist today.

4 Regional Training and Partnerships

Demonstrating applications of the NASA-LCLUC research and providing training in remote sensing analysis to local participants is extremely important for building local capability to continue the land cover monitoring work. The NASA-LCLUC projects in the region have engaged local scientists and developed partnerships with government and non-government organizations (NGOs) in the conduct of the research. For example, after two training sessions and working closely with the University of Maine and NASA researchers on change detection methodology, CONAP, (the Guatemalan government protected areas agency) working with a local NGO (ProPeten) in Flores Guatemala, successfully completed the 2001 forest change mapping update for the MBR (Ramos Ortiz and Martinez, 2001). The SYPR project has collaborative research linkages with El Colegio de La Frontera Sur (ECOSUR), a Mexican research and teaching institute focused on issues of forest conservation and sustainable development at Mexico’s southern frontier. The project has also disseminated research findings and maps to the Calakmul Biosphere Reserve, NGOs such as Bosque Modelo Calakmul, the regional council of ejidos, as well as in individual ejido communities. Also, the Mesoamerican Biological Corridor project has conducted five training workshops in four different Central America countries to build local capacity to use NASA technology in monitoring LCLUC in the region.

An exemplary feat of regional cooperation was achieved when Conservation International spearheaded the Selva Maya Vegetation Mapping Project, that leveraged several land cover mapping projects ongoing in the forests connecting the Mexican states, northern Guatemala and Belize. The vegetation maps for the Maya Biosphere Reserve in Guatemala (Sader et al., 1997), the Calakmul Reserve in Campeche and part of Quintana Roo, Mexico (Turner et al., 2003) were contributed by the two NASA-LCLUC funded projects previously mentioned. Other contributors to the Selva Maya map were 1) ECOSUR supplying the data for the Monte Azules Biosphere Reserve and environs in Chiapas, Mexico; 2) Stanford University’s Center for Conservation...
FOREST CHANGE AND HUMAN DRIVING FORCES

Biology, contributing data for part of the Calakmul Reserve and; 3) the Programme for Belize and Ministry of Natural Resources, providing complete vegetation maps for Belize.

The technical challenge of the Selva Maya project was to present a consistent and seamless vegetation map, given that different classification schemes, image resolutions and land cover interpretations were used to develop the original map products. Each contributing institution “crosswalked” their respective vegetation classifications into the Federal Geographic Data Classification – National Vegetation Classification System. The Selva Maya vegetation map was published by Conservation International in May 2000 (www.conservation.org) to facilitate regional cooperation and integration of geographic data bases to support sustainable development and biodiversity conservation in the region.

5 Conclusions

The NASA – LCLUC research in the region has demonstrated the utility of medium resolution satellite imagery (e.g., Landsat) for monitoring forest change. Furthermore, the analysis of remotely sensed data combined with the collection and examination of socio-economic and biophysical data has improved our understanding of the driving forces that influence human decisions to clear forest, primarily for agricultural uses, in the Central American region.

The use of corridors to link larger protected areas, is receiving growing endorsement by conservation biologists as a strategy to foster biological connectivity (and reduce local extinctions) in a practical and ethical manner, considering that large protected areas cannot always be set aside in areas where people have lived and farmed for centuries (Bennett, 1999). Ideally, areas within the proposed corridor links would be allowed to regenerate back to forest and managed for “green” land management activities (e.g. diverse cropping, shaded coffee, riparian forest protection, etc.) that could be promoted and encouraged through financial incentives as part of an MBC strategy (Miller et al., 2001). Although these ideas sound good and reasonable on paper, accomplishing these sustainable development goals are extremely challenging. There are many complex social, cultural, and technical issues that have to be carefully considered and resolved to implement multiple projects on the ground that will lead to improved linkage of forest and woody habitat corridors throughout the region. Another 30 million dollars has been pledged by international donors in 2003 to continue support for MBC projects (ccad.sgsiga.org).

In a comprehensive report on the MBC development strategy (Miller et al., 2001), the World Resources Institute (WRI) stated that “the MBC’s success will depend on the collection and dissemination of accurate, relevant and appropriate information to the broad array of decision-makers and stakeholders involved”. Adequate GIS databases, compatible for the entire region, and including a current, high-resolution land cover/use map still does not exist to support detailed analysis of the entire MBC. However, high resolution satellite images are now available (complete 1990s and 2000 Landsat data -30 m) through the NASA Scientific Data Purchase program which has stimulated new cooperative research between NASA, the World Bank, U.S. Agency for International Development (USAID) and university scientists to push the research agenda focused on biodiversity analysis of the MBC and regional carbon initiatives.
6 References


Introduction

Human beings are altering land cover at rates and scales that are unprecedented in human history (NRC 2002), rivaling glacial/interglacial transitions in magnitude (NAS 2000). Nowhere are human-mediated changes in land cover affecting global processes more than in the tropics. Understanding the causes and effects of regional land cover and land use change (LCLUC) is one of the grand challenges in the environmental sciences (NAS 2000). Further refinement in the estimates of tropical forest conversion will also be important for balancing the global carbon budget, and reconciling flux estimates from models (Houghton et al., 2000) and atmospheric measurements (Ciais et al., 1995a, 1995b).

Land cover and land use change are important drivers of ecological change in Amazonia. Conversion to agricultural and urban land creates widespread ecological disturbance, even at some distance from the zone of direct encroachment (Walker and Solecki, 1999). Land cover change in this region is globally significant, having a large influence on hydrology, climate, and global biogeochemical cycles (Crutzen and Andreae, 1990; Houghton and Skole, 1990; Salati and Vose, 1984; Shukla et al., 1990; Houghton 1991).

Despite the recognized importance of tropical LCLUC, which affects everything from aerosols and biodiversity to the global carbon and hydrologic cycles, hard data on LCLUC in these regions have been sparse or nonexistent. Consequently, global change scientists have had to rely on frequently inaccurate FAO estimates of forest loss (Kaiser 2002). This has forced the Intergovernmental Panel on Climate Change (IPCC) to emphasize that deforestation estimates in tropical countries may be in error by +/- 50%. Past FAO estimates have suffered from a strong reliance of secondary sources, or very sparse sampling; the comprehensive use of remote sensing data has been heretofore absent in the implementation of the observation and measurement efforts, and only recently considered for future assessments. In recent years, several important LCLUC studies of tropical forest loss or degradation have been published in the scientific literature, some utilizing remote sensing observations (cf. Nepstad et al., 1999; Achard et al., 2002; DeFries et al., 2002;), however, these studies, when taken together as a whole, contain considerable uncertainty and broad differences in our current understanding of rates of deforestation and degradation. Indeed, all of these studies have had to rely on indirect estimation techniques, or imperfect sampling schemes to make large-area or global estimates.
Provision of accurate and timely land cover and land use change information is problematic for a number of reasons, not the least of which is the near 2 billion hectare scale of the Earth’s humid tropical forests. Recent efforts to estimate global tropical deforestation rates (Achard et al., 2002; DeFries et al., 2002) have only served to illustrate the problems faced. Although both studies indicate that FAO estimates are high, they differ widely on the continental estimates of where the errors are located. What is widely agreed on is the need for wall-to-wall, high-resolution forest/deforestation maps in the tropics (Achard et al., 2002; Kaiser 2002; Schimel and Baker, 2002; UNEP 2002).

Despite the numerous environmental and ecological effects across all scales of analysis, most of the analysis of land use and land cover change has been coarse and highly aggregate, with a restricted focus on its role in global phenomena, particularly the carbon cycle. The use of remote sensing affords the opportunity to obtain focused disaggregated measurements of land cover changes in key regions such as Amazonia. Over the past decade, considerable work has been done using remote sensing to measure the rate and pattern of deforestation in this important region. But there are serious limitations to previous analyses. First, we lack accurate measurements of the inter-annual variations in the rate and geographic extent of deforestation and we do not know the significance of inter-annual variability on fluxes in water, energy, carbon, and trace gases. Second, we do not know the magnitude and importance of forest regeneration, the area of which is quite variable from year to year. Moreover, we do not well understand the overall dynamics of secondary growth, including its probability distribution function from one region of Amazonia to the next, how fast it regrows, how long it persists before being re-cleared, how often it is recycled, or how land management affects these processes. Lastly, we have extremely poor information on forms of forest degradation which occur in addition to outright loss of forests by deforestation; degradation occurs through selective logging, fragmentation and fire.

This paper reviews our current understanding of the rates and geographic distribution of the full disturbance regime in Amazonia. The objective of the research reported here is to provide a 30-m resolution multi-temporal regional measurement of deforestation, regeneration, and degradation in Amazonia. We consider the recent trends in deforestation and degradation and utilize remote sensing to measure forest loss rates, regeneration, and logging. While previous work under the Landsat Pathfinder program has emphasized forest and non forest mapping (i.e. deforestation) we now include measures of a more complete suite of disturbances including logging as well as regeneration of forests. Among other things, this study makes a definitive statement on the quantitative significance of degradation via logging, and an assessment of recent studies which have used indirect estimators to claim logging rates within tropical forests as high as or higher than deforestation (e.g. Nepstad et al., 1999).

2 The Science of Amazon Forest Cover Change

2.1 TROPICAL LAND COVER AND LAND USE CHANGE

The world’s biodiversity-rich tropical forests house the majority of the Earth’s species. These forests also, by virtue of their high biomass, hold great quantities of carbon. Therefore, understanding the spatial extent and underlying processes inherent in rapid
tropical LCLUC is critical for a host of reasons, from the conservation of habitat, resources, biodiversity and environmental services, to concerns about trace gases and particulate emissions that alter climate and impact human health.

A byproduct of these land use derived cover changes is extensive forest degradation and fragmentation that alters disturbance regimes for hundreds to thousands of meters from forest edges due to biomass collapse, fire and other secondary effects (Laurance et al., 1997; Laurance et al., 2000; Cochrane 2001; Cochrane and Laurance, 2002), thereby degrading vast regions of forest beyond the areas of outright clearing (Skole and Tucker, 1993; Skole et al., in prep). In addition, tropical forests throughout Africa, Asia and the Americas continue to be subjected to large-scale selective logging operations that reduce the structural and biophysical integrity of millions of hectares of forest each year (Nepstad et al., 1999; Barber and Schweithelm, 2000; Matricardi et al., 2001; Laporte et al., in press). Both fragmentation (Cochrane 2001) and selective logging (Cochrane et al., in press) interact synergistically with fire-dependent land use to foster frequent and increasingly intense forest fires (Cochrane et al., 1999) that have degraded several million hectares of tropical forest in recent years (UNEP 2002, Cochrane 2003).

2.2 FOREST DEGRADATION

Understanding changes in tropical land cover/use requires more than simple quantification of deforestation rates because there are also a range of processes which degrade forests, such as forest fires, forest fragmentation and selective logging; these processes reduce canopy cover and cause biomass to be lost from extensive standing forests, even while forest cover nominally remains intact. Achard et al. (2002) estimated, through visual interpretation, that 2.3 million ha yr\(^{-1}\) of standing tropical forests were degraded between 1990-1997. This estimate is obviously low given that by 1999, 10.3 million hectares of forest edges (Skole et al., in prep) were suffering biomass collapse (Laurance et al., 1997) and over 2.3 million hectares were selectively logged in the Brazilian Amazon alone (Matricardi et al., 2001). Pan-tropical forest fires burned more than 20 million hectares in 1997-98 (Cochrane 2003) including more than one million hectares of Amazonian forests in the Brazilian state of Roraima (UNEP 2002). Incorporation of forest degradation into LCLUC analyses will require moving beyond discrete land cover classification to accurate continuous-field representations of the spatial and temporal changes in forest density and structure that describe the degradation-regeneration continuum of forest cover.

Forest degradation affects millions of hectares of forest. However, unlike deforestation, degradation, even if severe, may only be apparent in standard imagery classification techniques for a few years due the rapid regrowth of vegetation that quickly masks the damaged forest (Stone and Lefebvre, 1998; Souza and Barreto, 2000). Recent advances in methods for both indirect (Logging – Souza and Barreto, 2000; Janeczek 1999; Matricardi et al., 2001; and fragmentation – Skole and Tucker, 1993; Skole et al., in prep) and direct measurement (fire-damaged forests – Cochrane and Souza, 1998; Siegert and Ruecker, 2000) of forest degradation have advanced our understanding of the magnitudes of these disturbances.

Despite these advances though, accurate measurements have yet to be made. In some cases, studies have relied on indirect measures. For example, Nepstad et al. (1999) extrapolated the area of forest subjected to logging from the volume of roundwood arriving at sawmills. The problem with such analyses is that, without mapping, serious
double counting of land cover change may occur. Simple extrapolations, to estimate
degraded forest area, which lack explicit connection to observed changes in land cover
or use, cannot account for spatial or temporal overlap with other land use or cover
changes. In this particular case, there was no accounting for roundwood arriving at
sawmills that originated from newly deforested lands (salvage of valuable trees) or
from sites that had previously been logged (return logging for newly marketable
species). These activities are important in the Amazon but vary spatially and temporally
across the Amazon basin. Hence, the most widely reported estimate of logging rates in
virgin forests of the Amazon (Nepstad et al., 1999) may be considerably flawed as a
measure of damage to standing forests (“cryptic impoverishment”) because the
investigators could not explicitly map these disturbances on the landscape.

2.3 SIGNIFICANCE OF DEFORESTATION FOR CARBON ACCOUNTING

Deforestation processes have been well documented in tropical forests (Geist and
Lambin, 2002). However, considerable uncertainty remains regarding the complex
mosaic of damaged and regenerating forests that surrounds it. Estimated changes in
terrestrial carbon stocks that are caused by land use change in the tropics often use rigid
binary classification techniques (deforestation assessment) where a pixel is classified as
either forest or non-forest. This approach cannot account for land use effects and other
disturbances that reduce the carbon content within standing forests. Even if a pixel is
classified as forest, there can be substantial differences among forests in terms of
carbon stocks, due to fractional density of forest cover and the ages of regrowing trees.

Improved carbon models will require remote sensing measurements that account
for fractional changes within forests as additional source terms, such as those caused by
logging and fire (Cochrane et al., 1999; Nepstad et al., 1999) or addition sink terms such
as regeneration (Brown et al., 2000). There is widespread recognition of the importance
of including biotic components in global carbon models, which have typically introduced
land use change as a simple forcing function as aggregate totals of area cleared on
decadal time steps, with little consideration of spatiality. Parameters used to drive these
models are often derived from surrogate data, such as agricultural land areas (e.g.
Houghton et al., 1999). Most current efforts to incorporate land use change into carbon
cycle models have been aggregate in construct (0.5° - 4.0° spatial resolution). These
models have not explicitly treated the spatially varying landscape mosaic left behind by a
range of anthropogenic disturbances acting simultaneously and interactively within a
locale. As a consequence, much of the research and modeling is undertaken at a scale that
prevents a comprehensive accounting for land use effects from a range of disturbances,
including degradation and regeneration within intact forest covers. Nor do these simple
trend models or datasets-derived analyses (e.g. Ramankutty and Foley, 1999) provide
prognostic insight into actual processes associated with land use and land cover change.

The distribution of sources and sinks of carbon among the world’s ecosystems is
uncertain. Some analyses show northern mid-latitude land to be a large sink, whereas the
tropics are a net source (Ciais et al., 1995; Fan et al., 1998). Other analyses show the
tropics to be nearly neutral, whereas the northern mid-latitudes are a small sink. (Keeling
et al., 1996; Rayner et al., 1999). The role of land use change is central to this
controversy, since the so-called missing sink of carbon is either a component of changes
in ecosystem metabolism or secondary growth from recent land use change. It will be
difficult to determine through direct measurements if it is due to ecosystem metabolism,
but remote sensing of changes in forest are a useful way to measure the land use component and either confirm or eliminate this component. Also, even if ecosystem metabolism were a major component, its magnitude would have to be estimated from, in part, knowledge of the magnitude of the tropical land use source.

Studies of tropical land use change suggest considerable inter-annual variation in rates of deforestation (Skole et al., 1998; Alves and Skole, 1996; Steininger 1996; Moran et al., 1994). Moreover, there are large areas of formerly deforested land being placed into secondary succession each year (Skole et al., 1994). In specific years when deforestation rates are low and abandonment is high, this might provide brief periods when carbon sources and sinks from land use change are nearly balanced. Houghton et al. (2000) investigated annual fluxes of carbon from deforestation and regrowth in Amazonia, the single largest tropical forest zone, and concluded that carbon emissions from land use change were balanced by carbon uptake in undisturbed forests (Tian et al., 2000), but regrowth on recently deforested land was not large enough to suggest land use changes alone could result in a neutral net flux. However, this analysis showed considerable inter-annual variation and only used a single year’s estimate of the area in regrowth. Thus, it could not assess through direct observations (cf. from remote sensing), inter-annual variations in regrowth compared to deforestation explicitly. Moreover, the importance of selective logging and forest degradation through fragmentation and fire were not included in that analysis.

Thus, there is substantial indirect evidence from historical land use records, atmospheric measurements, and timber volume records, that land use change results in large net fluxes to the atmosphere in the tropics, and is responsible for a component of the missing sink in the northern temperate zone. There is less direct evidence from remote sensing of changes in land areas, particularly on an annual basis. This is true for the tropical regions where there are on-going efforts to measure deforestation and regeneration, but not comprehensively including other forms of human-caused forest disturbance and degradation on an annual basis.

3 Geographic and Inter-annual Variations: Recent Evidence for Land Use Components

Early work by Tans et al. (1990) used a global circulation model to redistribute atmospheric CO₂ based on surface fluxes of CO from fossil fuel combustion, tropical deforestation, and oceanic uptake. Their analysis predicted that atmospheric [CO₂] should exhibit a strong north-south gradient of 5.7 to 7.3 ppm. This estimate was significantly higher that the observed meridional gradient of 3ppm based on the NOAA flask measurements (Tans et al., 1990). Therefore, they assumed that the “missing sink” was a terrestrial sink of 2.0 to 3.4 GTC in the northern hemisphere temperate zone. Siegenthaler and Sarmiento (1993) updated the Tans et al. oceanic uptake estimates, and concluded that there is either a terrestrial sink of about 1.8±1.3 GTC, or that the tropical source term of 1.6±1.0 GTC is too high, or a combination of both. Quay et al. (1992) estimated the net flux of CO₂ in the ocean and biosphere using isotopic measurements of δ¹³C (essentially the ratio of ¹³C/¹²C) in the atmosphere and dissolved inorganic carbon from ocean surface waters and atmospheric [CO₂], and estimated that the average annual terrestrial uptake was negligible (0.1GTC).
More recently Ciais et al. (1995a, 1995b) used a hybrid approach using a two-dimensional atmospheric transport model (like Tans et al., 1990) with [CO$_2$] and $\delta^{13}$C measurements from NOAA’s Climate Monitoring and Diagnostics Laboratory (CMDL) global flask network (like Quay et al., 1992) to estimate CO$_2$ partitioning as a function of latitude and time from 1990 to 1993. Their results suggest that the northern temperate zone was a large terrestrial sink (3.5 GTC) in 1992 and 1993, whereas the biosphere in the northern tropics (from equator to 30°N) was a large source (2.0±1.3 GTC) of carbon.

They also estimated that the southern tropical zone was a small terrestrial sink in 1992 and 1993. While a northern tropical source is consistent with satellite measurements of deforestation and fires in Indochina (Skole et al., 1998), this is surprising for southern tropics because this latitudinal band contains Brazil and Indonesia, two areas with very high reported rates of deforestation. A weak terrestrial sink in the southern tropical zone suggests that either the forests have a positive net ecosystem production (due to fertilization of undisturbed forest and/or secondary growth formation) or previous estimates of deforestation have been overestimated (Ciais et al., 1995a). Keeling et al. (1996) provide particularly strong evidence for a neutral tropical flux from new measurements of O$_2$.

Houghton et al. (2000) modeled both the annual variation in Amazonian deforestation, a large southern tropical zone, and changes in NPP due to climate variations in Amazonia based on Tian et al. (2000) and concluded that while deforestation was a large source, it was offset by increased NPP in undisturbed forests. However, there was considerable variation from year to year in both components (land use and ecosystem metabolism) of the regional total net flux.

4 Limitations of Global Models and Their Treatment of Land Use

Many previous efforts to model forest and ecosystem changes have been aggregate in nature (0.5° - 4.0°), and used to study primarily biophysical aspects: biogeochemistry (e.g. carbon flux), changes in ecosystem metabolism (e.g. NPP) or changes in water and energy balance (e.g. albedo or latent and sensible heat flux), without specific or comprehensive regard for the complex dynamics of land use itself. For instance McGuire et al. (1997), Tian et al. (1998) and Melillo et al. (1993) have used TEM, a large scale ecosystem model, to explore ecosystem response to changes in temperature and precipitation due to increased atmospheric CO$_2$ and radiative forcing on climate. Potter et al. (1996) have used the CASA model to explore continental scale hydrology and carbon dynamics. The SiB models have been used to examine changes in surface conditions and ecosystem productivity, and have been coupled to large-scale climate models to provide more realistic vegetation and land cover controls on climate dynamics. Similarly the SVAT and BATS models have been employed in the aggregate to improve land cover factors in climate models. At still more aggregate levels, the constrained inverse models (cf. Randerson et al., 1997; Tans et al., 1990) have been used to examine sources and sinks of carbon at the global scale, with spatial heterogeneity limited to latitudinal gradients of atmospheric CO$_2$.

Global ecosystem models have typically introduced land cover change as a simple forcing function derivative of total area cleared. For example, most models use aggregated deforestation rates to drive carbon flux or changes in continental NPP. When a spatial context is introduced it has generally been done to treat the large-area
variations in biomass classes found on generalized maps or for connectivity to basin-scale hydrology. These models have been invaluable for ascertaining the global-scale significance of land use and land cover change for forcing fluxes of radiatively important gases, climate forcing from land surface changes, or biome-wide ecosystem response to climate change. The scale selected has been consistent with the questions they pose. The treatment of land use change has also been consistent with their approach, primarily focused on developing inventory-based calculations as a direct function of the total area disturbed.

This generalized picture of global change, with respect to land use and cover change, is far from complete. These models have not explicitly treated the spatially varying landscape mosaic left behind by a range of anthropogenic and natural disturbances acting simultaneously and interactively within a Disturbance Regime. As a consequence, much of the research and modeling is undertaken at a scale that overlooks actual processes associated with land use and land cover change.

Within the present context, the following four shortcomings of an aggregate modeling approach can be identified:

4.1 INTERACTIONS BETWEEN MULTIPLE DRIVERS OF LAND COVER CHANGE

The first point is that a fundamental understanding of the processes of land use and land cover change is absent, and therefore any prognostic land use capabilities are non-existent in global models. A time series of deforestation, even represented as a coarse scale map, does not reveal the actual and multiple factors that are causing (or are driving) the patterns and rates, because the factors at work occur and vary across scales, and exist as a suite of factors, or agents, operating in a single landscape and interacting with each other.

Consider the range of disturbances in a tropical forest, which includes agricultural deforestation, logging, fire, and plant pests and pathogens. In Amazonia, selective logging and the construction of logging roads lead to deforestation for agriculture (Verissimo et al., 1995). Selective logging degrades forests, resulting in local drying of these sites (Uhl and Kauffman, 1990). The use of fire in shifting cultivation increases the probability of devastating fires in nearby logged forests (see Nepstad et al., 1999; Cochrane et al., 1999), and stimulates invasions by plant pests and pathogens. Fire probability in logged sites is increased by the actions of farmers clearing proximate forests for agriculture (Cochrane et al., in press). Both forest fragmentation and selective logging combine to synergize the spread of fires across the landscape and facilitate its penetration into remnant forests (Cochrane 2001; Cochrane and Laurance, 2002; Cochrane et al., in press). The risk of a fire “externality” in landscape-level farming systems tends to push farmers to adopt fire resistant pastures, a land use strategy that minimizes exposure to fire-related losses that are inherent in such long-term investments as agroforestry, thus greatly influencing the future trajectory of land use change in a region. Moreover, beyond the discrete boundaries of transition from “human” domination to nature, there is a gradient of biological impoverishment extending into what is generally identified as “primary” forest. As this example makes clear, land use and land cover change (in the tropics and elsewhere) arises by virtue of complex interactions, leads to unexpected feedbacks, and broadcasts ecological impact beyond the boundaries of human use, and abuse, of land.
4.2 FINE–SCALE SPATIAL PATTERNS

Second, the pattern of landscape change induced by land use change is revealed at very fine scales. The landscape that results from the suite of agents of land use change – both human and biophysical in origin – is complex and not well described in the aggregate. To be sure, recent studies by Laurance et al (1997) have shown significant biomass collapse in the forest edges adjacent to cleared pastures due to changes in microclimate, among other factors, which result in increased tree mortality and decreased recruitment. The influence of the landscape mosaic or patchwork resulting from various modes of disturbance and recovery influences water and energy balance, latent and sensible heat flux, as well as atmospheric boundary layer dynamics. Proximity, size, and spatial orientation of the landscape matrix matter for the prediction of land cover change (Gascon et al., 2000; Cochrane and Laurance, 2002).

4.3 FINE–SCALE TEMPORAL PATTERNS

There is considerable evidence that disturbance to forests by human activities varies significantly from one year to the next. Deforestation rates in Amazonia, for example, may range by a factor of two from year to year. While most models use simple linear extrapolations of forest clearing as a forcing function, usually aggregated or averaged on a decadal time frame, we know that the fine scale temporal patterns of clearing, regeneration from abandonment or fallowing interact in counter-intuitive and complex ways (Skole et al., 1998). Satellite analyses of the Brazilian Amazon indicate that from 1978 to 1998 the average annual rate of deforestation was between 1.6-2.2 x106 ha per year (Fearnside 1993; Skole and Tucker, 1993; INPE 2002). The 1990-1991 rate was 67% less than the average rate for this 20-year period.

From a few site studies in which interannual time series data have been acquired, it has been shown that the abandonment rate can be as high as 82% of the deforestation rate (1991-1992 in Alves and Skole, 1996). During this same period, the deforestation rate varied annually by as much as 100% with an average variation, over 6 years, of 50%. Satellite analysis indicate that, in parts of Rondonia, secondary growth is a large (over 40% of the deforested areas) and a rapidly changing pool. Almost 60% of the areas in secondary growth in 1986 were reclared at least once by 1992, and over 55% of the deforested areas in 1986 were abandoned into secondary growth for some period of time by 1992 (Alves and Skole, 1996). This analysis and other similar satellite-based analyses (Lucas et al., 1993; Brondizio et al., 1994; Moran et al., 1994; Steininger 1996) are useful for characterizing the dynamics of the secondary growth resulting from tropical land use.

These and other recent studies of deforestation from satellite data show a highly dynamic process of clearing, abandonment and re-clearing, and that the rates at which land is cleared or abandoned are related to the land use and management system that forest farmers employ. In some cases, deforestation and secondary succession exist in tandem as a tightly coupled system in which secondary growth is continually recycled back into farmland. In other cases, active land management maintains the land in agriculture, or the lack of active land management or population displacement results in long-term succession.
These issues then beg a very interesting suite of LUCC questions concerning the factors that cause deforestation rates to vary from year to year, and what factors determine or control the balance between clearing and abandonment. The resolution of such questions depends completely on: (1) making fine temporal resolution measurements (annually) of deforestation rates over large areas using satellite remote sensing, (2) making fine spatial and temporal scale analyses of the dynamics of land use and cover change in order to document if regrowth is a significant factor, and (3) developing an improved quantitative and diagnostic capability to determine the factors (ergo the so-called Human Dimensions) which control rates of clearing and regrowth.

4.4 CHANGES INDUCED WITHIN COVER CLASSES WITHOUT RESULTING IN OUTRIGHT TRANSFORMATION OR CONVERSION

Fourth, the use of general classes of land cover as homogeneous classes masks the fact that gradients exist for important biophysical attributes and that boundaries between classes are not abrupt, either between different natural covers or between natural and disturbed covers. Moreover, alteration within forests, through extraction or degradations (e.g. selective logging or burning) is important to human use of forest natural capital. In addition, forest fragmentation related disturbance is often larger in area than forest loss in any given region (Skole and Tucker, 1993). Laurance et al. (1997) showed that biomass collapse along forest edges follows a linear gradient from the pasture to 100 meters into the forest with increased selective mortality of large trees out to 300 meters (Laurance et al., 2000). Escaped fires burn the understory of neighboring forest for thousands of meters (Cochrane 2001), while selective logging and other human actions degrade intact forests but do not result immediately in an outright conversion (Uhl et al., 1997). Such changes are highly significant to ecosystem structure and function, therefore, sharply drawn boundaries may represent a critical loss of information. The landscape at any given time is not simply a collection of land covers but may represent a legacy of previous disturbances that can alter future land use and cover trajectories. For example, forest burning can result in a positive feedback wherein subsequent fires become both more frequent and severe until complete deforestation results (Cochrane et al., 1999). Moreover, spatial relationships, or topological relationships, between land use and land cover types influence both the dynamics of change and ecosystem impacts. Perennials, plantations and pastures exhibit vast differences in their effects on neighboring ecosystems.

5 Multiple Facets of Land Cover Change: An Example Situation from Amazonia

Simple, binary, forest-nonforest representations of the tropics miss the rich detail of forest degradation that exists and also many of the fundamental drivers that are changing these landscapes. There is a linkage between land conversion, selective logging and forest fire in many regions of the tropics. Fire-maintained agricultural activities, often supported by or in close proximity to selective logging operations, fragment a region’s remaining forests and can result in escaped fires moving into standing forests. Logging opens the canopy and allows these forests to rapidly lose moisture and dry out. Many forests are revisited several times, when loggers return to harvest additional lucrative tree species as regional timber markets develop (Uhl et al.,
1997; Veríssimo and Amaral, 1998). These forests become very degraded and may have 40 – 50% of their canopy cover destroyed (Uhl and Vieira, 1989; Veríssimo et al., 1992). Landscape fragmentation and land cover change interact synergistically to expose more of the forest to fire and consequently raise the risk of unintended fires occurring across the entire landscape (Cochrane 2001; Cochrane and Laurance, 2002).

As selective logging and periodic forest fires occur, the landscape matrix, which was formerly made up of flammable islands of vegetation surrounded by green, forested firebreaks, becomes increasingly porous, allowing fire contagion to spread more easily across the landscape. This can initiate a positive feedback of increasing fire susceptibility (Cochrane and Schulze, 1999), increasing fuel loads and increasing fire severity (Cochrane et al., 1999). Return logging and fire can combine to dramatically change forest structure, reduce on-site biomass and lead to extensive invasion of vines and grasses (Uhl and Kauffman, 1990; Veríssimo et al., 1992; Cochrane and Schulze, 1999). At present, thousands of square kilometers of tropical forests are logged each year in the tropics (Nepstad et al., 1999; Janeczek 1999; Barber and Schweithelm, 2000; Matricardi 2003). Logging and fire damage have combined to create a vast pool of tropical forests that are neither intact nor completely destroyed but which may have 10-80% reduced on-site live biomass/carbon storage (Uhl et al., 1997; Cochrane and Schulze, 1999; Cochrane 2003).

6 An Opportunity to Develop New Remote Sensing Methods

Given the concern resulting from the high global rates of deforestation and the largely unquantified rates of forest degradation in the tropics, there has been much interest in quantifying, mapping and understanding the processes of land cover and land use change. Due to the sheer size of the area involved, the use of remotely sensed satellite data is the only practical approach for studying these changes over large areas. Coarse-resolution data are manageable and can provide some information about land cover change in the tropics but it has become increasingly apparent that many of the changes and actual LCLUC processes occur at finer scales. The need for quality high-resolution land cover information is critical the world over. To address the perceived needs of the science community, NASA is currently compiling a dataset of Landsat 7 ETM+ imagery for the entire globe for the year 2000.

Although annual, wall-to-wall, high-resolution mapping would be ideal, the logistical, technical and financial challenges make it impractical at the present time. Therefore, estimation of LCLUC through spatial sub-sampling has been put forth as a reasonable alternative. The question however is, how should the sampling be conducted and weighted in order to provide accurate LCLUC estimates? Skole et al. (1997) showed that random sampling was not appropriate because LCLUC was not randomly distributed and put forth a targeted sampling scheme based on data from existing high-resolution maps of the Amazon. In the Amazon, this approach of using a select subset of known important imagery to estimate deforestation for the entire region has been used as rapid evaluation tool and precursor to complete basin-wide coverage for several years now (INPE 2002). More recently, Achard et al. (2002) have used a pan-tropical sampling, based on previous deforestation maps and key informant input, to estimate LCLUC. Now, the FAO Forest Resource Assessment (FRA) is proposing to use a stratified sampling method that is strictly geographic, without use of any region-specific information. At
present, there is no way of evaluating the utility or accuracy of these sampling schemes since there is no ‘known’ land cover change “population set” to which the samples are being compared.

7 Measurement of Deforestation and Degradation in Amazonia

We utilized the Landsat series of satellites to measure the area and location of deforestation, regrowth and selective logging throughout the Amazon basin in Brazil for the years 1992, 1996, and 1999 (2001 is forthcoming). These measurements provided digital maps of the extent of deforestation, regrowth and logging and are being distributed on the world wide web. In this section of the paper we describe our approach and results. As far as we know, these are the first and only comprehensive assessments of the full suite of disturbances in the basin.

7.1 METHOD: DEFORESTATION, REGENERATION, AND LOGGING MAPPING

The entire Legal Amazon was mapped using Landsat Thematic Mapper and Enhance Thematic Mapper data. Approximately 220 individual Landsat scenes were acquired to cover the entire region for the benchmark years 1992, 1996 and 1999. All images acquired were as close to these dates as possible, given cloud cover problems, and the majority of images used were on or within one year of the benchmark years. These data were georeferenced by co-registration to the Earthsat GeoCover database, which is a collection of data for the mid 1990s purchased by NASA under the Data Purchase Program directed at the Stennis Space Center. Registration to true earth coordinates was within 80 meters. Each image was processed by simple unsupervised classification clustering techniques (e.g. ISODATA in the ERDAS image processing software), and then reduced to 7 distinct classes: forest, cerrado, non-forest/deforested, regeneration, clouds, cloud shadow, and water. Individual classified images were then merged to form a seamless map for the entire region. This process was repeated for each benchmark year and simple summary statistics were tabulated which provided an inventory of the total area deforested and the total area in regeneration for each year. Rates of net deforestation were tabulated by overlay comparison of each year as separate data layers in a geographic information system. This post-classification change analysis produces average annual rates of deforestation between benchmark years as well as an inventory of regenerating forest at each year.

Logging detection requires a technique which maps changes within the forest class and is, thus, different than that performed for deforestation mapping. The original data used for the deforestation mapping described above were first visually inspected at full resolution for evidence of possible logging operations. It is important to realize that logging in these forests is normally very selective with only a few of the most lucrative species and larger trees being extracted. This logging activity frequently leaves behind visible roads and log storage patios. Detailed site and ground studies reveal that most areas with logging intensities above 10 cubic meters per hectare are detectable (Matricardi 2003). Additional tests were performed to evaluate the efficacy of visual screening compared to radiometric measurements using automatic methods, following the methods described by Janeczek (1999). Scenes where logging activity was suspected were removed from the full dataset for further analysis. The analysis followed the technique
described by Janeczek (1999) and Matricardi (2003). First, areas of logging were evaluated by digital image processing using a spectral texture analysis technique (Matricardi 2003) that identified areas of significant logging by detecting the specific location of logging patios, roads, and other indicators of logging activity. We used the work of Souza and Barreto (2000), which showed that trees extracted during logging operations came from a radius of 180 meters around logging patios, to delineate the buffer area around logging infrastructure which was logged, and hence, degraded by logging. Field sampling by our team confirmed that this approach was valid for the areas with the most intense logging which account for the vast majority of Amazonian timber extraction. In areas of less intense, more selective, logging, patios were found to be more widely spaced and buffer distances from which trees were drawn were found to exceed 400 meters at times (Matricardi 2003). Subsequent to the automated detection, detailed visual analysis was conducted as well. Logging areas were visually traced using the full resolution image and heads-up digitizing. Intercomparison of the techniques using multitemporal analyses of series of annual images for case study sites showed that the methods had different strengths and both were conservative. The automated technique was excellent at detecting newer logging areas and smaller regions of logging activity but was increasingly poor at detecting logging patios from logging that was older than 1 year due to rapid forest regrowth. Conversely, the visual detection was less precise at delineating the actual areas logged and missed many of the smaller regions of logging but it was better at detection of older areas of logging activity. If annual imagery for a site exists then the automated technique would be sufficient for tracking logging activity but in areas where multiple years between images occur then a combined automated and visual approach is best.

The data layers developed from the visual, textural and buffer analyses were merged for each benchmark year. Areas were also compared with the maps of deforestation and regeneration to ensure that any plot of land was classified as only one type disturbance in order to prevent any double counting. Logging and deforestation/regeneration datasets were then merged to produce a composite result.

8 Results

Figure 1 shows a map of the current areas of forest and deforestation in Amazonia. Figure 2 shows the map of current areas under logging. Table 1 presents summary results from this analysis, indicating the total area deforested, in a state of regeneration, and under logging for each of the three benchmark years. The total area deforested increased from 332,563 km$^2$ in 1992 to 399,445 km$^2$ in 1996 and 485,686 km$^2$ in 1999. These data extend the results of Skole and Tucker (1993) which indicated the total area deforested in 1988 was 230,324 km$^2$. Hence, the total area deforested has more than doubled since the last reporting in the literature. Our results suggest approximately 13% of the forested portion of the Amazon Legal has been converted to deforested land. Of the area deforested approximately 30-40% (30%, 35%, 40% for 1992, 1996, 1999 respectively) is in some stage of regeneration. Preliminary work and previous analyses suggested that while this pool size of regeneration remains rather stable, it is rapidly turning over; areas in regeneration may stay in that class only for 2-5 years before they are reconverted to the deforested class.
Figure 1. Results from deforestation and regeneration mapping for 1999. Areas of deforestation are shown as white dots and outlines. Most of the deforestation is concentrated in areas of historical deforestation along a crescent along the southern fringe of the closed-canopy forest zone. Most intensive deforestation occurs in the states of Para, Mato Gross and Rondónia.

The average annual conversion rate was $28.7 \times 10^3 \text{ km}^2 \text{ yr}^{-1}$ for the period 1996 to 1999, and $16.70 \times 10^3 \text{ km}^2 \text{ yr}^{-1}$ for the period 1992 - 1996. Compared to estimated deforestation rates for the previous decade (Skole and Tucker, 1993), we note that, even while there is significant inter-annual variation, deforestation rates appear to have risen rapidly in recent years. The rates for the period 1996-1999 are somewhat higher than those being reported by the Brazilian space agency (INPE 2002), which reports deforestation rates of $13.2 - 17.4 \times 10^3 \text{ km}^2 \text{ yr}^{-1}$. There is, however, close agreement with their estimates for the period 1992-1996, although we are slightly lower than their average of $19.2 \times 10^3 \text{ km}^2 \text{ yr}^{-1}$.

The analyses show that the area of regenerating forest increased over the time period of analysis (Table 1). This suggests that the area of the Amazon acting as a sink for carbon during any given year has increased; although over time, regenerating vegetation on previous agriculture sites is usually re-cleared, generating a source that off-sets the temporary sink. The fact that the total area of ongoing forest regeneration is increasing suggests an increased land-use-dependent carbon sink which, although having little or no net effect on carbon sequestration, can serve to dampen regional carbon fluxes to the atmosphere in some years (Houghton et al., 2000). At present, the relative synchrony or asynchrony characterizing new deforestation and the re-clearing of regenerating forests is unknown. The increase over time in the relative amount of area in some form of regeneration, as compared to the total deforested area, is also interesting in that it may
portend ongoing changes in land use, human behavior and landscape level response to human activity. For example, increasing percentages of areas in regeneration may indicate decreased productivity of land yielding higher rates of abandonment or earlier shifting to fallow. In addition, since the early 1990s, there has been a dramatic increase in both logging and fire in Amazonian forests. Both of these forest degradations can change the spectral signature of forests to appear similar to regenerating forests (Cochrane et al., 1999). Therefore, in part, the increasing percentages of regeneration may reflect growing stocks of highly degraded forest within the Amazonian landscape (UNEP 2002).

Figure 2. Results from the mapping of areas of degradation by selective logging for the year 1999. The Landsat images in which logging was detected are shown shaded in gray. The individual locations of logging areas are shown as black outlines and dots. Most of the logging is occurring in regions where there is intensive deforestation in the state of Para, with additional logging in Mato Grosso. Rondonia had logging in 1999 but it was not detected in any previous years. In such cases, it is highly likely that logged areas are rapidly converted to deforestation and hence those areas get mapped in the deforestation mapping procedure.

The results from analysis of logging present an interesting picture and also portray the increasing magnitude of degradation in addition to outright deforestation. The total area subjected to selective logging is shown in Table 1. The total area under logging was relatively small in 1992, and at 6 x 10^3 km^2 was only ~2% the magnitude of the deforested area. It seems unlikely that logging was a major disturbance factor prior to 1992. Since 1992, the area under selective logging has increased markedly to 26.1 x 10^3 km^2, but remains a relatively small impact in terms of total area disturbed compared to the
total area cleared for pasture and other conversions from deforestation. Unlike deforested areas, logging sites can be ephemeral. If left undisturbed, they rapidly regenerate and after several years logging activity is not detectable using current satellite imagery. Our analysis indicates that as much as \( 56 \times 10^3 \) km\(^2\) may have been affected by logging over between 1992 and 1999. However, this is still only 12% of the total area deforested.

Table 1. Total deforested area, area in regeneration, and area under logging for the observation years 1992, 1996 and 1999 for the Brazilian Legal Amazon. Also shown are average annual rates of primary forest conversion to deforestation for the periods, 1992-1996 and 1996-1999. The annual rate of new logging in undisturbed primary forests is shown for each observation year. (units are \( 10^3 \) km\(^2\) and \( 10^3 \) km\(^2\) yr\(^{-1}\))

<table>
<thead>
<tr>
<th>Disturbance type</th>
<th>1992</th>
<th>1996</th>
<th>1999</th>
</tr>
</thead>
<tbody>
<tr>
<td>Deforested area</td>
<td>332.6</td>
<td>399.4</td>
<td>485.7</td>
</tr>
<tr>
<td>Regeneration area</td>
<td>104.3</td>
<td>162.9</td>
<td>169.0</td>
</tr>
<tr>
<td>Logging area</td>
<td>6.0</td>
<td>10.0</td>
<td>26.1</td>
</tr>
<tr>
<td>Rate of deforestation</td>
<td>16.7</td>
<td>28.7</td>
<td></td>
</tr>
<tr>
<td>Rate of new logging</td>
<td>3.8</td>
<td>5.0</td>
<td>11.9</td>
</tr>
</tbody>
</table>

The rate of selective logging in primary forests has increased dramatically in recent years. Areas producing logs may be used for several years, and may eventually or immediately be converted to permanent clearings by deforestation. It is important to know the actual area of primary forest that is disturbed each year by logging; some observers suggest this impact could be significant. Thus, we calculated the rate of new logging each year using map overlays, field studies and detailed inter-annual analyses in case study sites, as described in Matricardi (2003); these studies differentiate those detected logging areas which are new from those which are re-used between successive time periods. These results are shown in Table 1 and suggest that the current rate of new primary forest disturbed by selective logging is \( 12 \times 10^3 \) km\(^2\) yr\(^{-1}\), or approximately half (42%) the rate of deforestation. While not a significant disturbance prior to 1999, logging now represents roughly 30% of the total disturbance in Amazonian forests.

The implications are significant. At the present time, these two important factors interact synergistically to catalyze disturbance in the Amazon. Both logging and deforestation are driving land cover change. Moreover, both of these factors interact with fire and fragmentation to exacerbate the total magnitude of disturbance (Cochrane 2003). It is easy to see a future with considerably more disturbance than anytime in the past.

9 Concluding Remarks

We are beginning to gain a more complete picture of land cover change in Amazonia. The results reported here only treat three aspects of the overall problem of measuring change in this important region – deforestation, regeneration, and logging – but a new picture is emerging which should put to rest some erroneous conclusions and myths. There has already been considerable additional work on the role of fire, which can be added to the results reported here (Cochrane 2003). Actual mapping of the locations of forest burning and relation of the spatial aspects and interactions with other land uses has lead to an appreciation of the dynamic changes being caused by this growing disturbance in the
Amazon (Cochrane et al., 1999; Cochrane 2001, Cochrane 2003). The next round of research on fire will need to focus on exact mapping of the area burned and its annual variation. Ongoing work, extending from the results reported here, is focused on quantification of fragmentation, edge effects, and biomass collapse as another form of degradation. This work will be important additional information, since there is speculation that the area of edge with biomass collapse in Amazonia is important for estimating carbon budgets (Laurance et al., 1998). However, experience from this analysis has shown that prior claims that logging was a large and unaccounted carbon flux were largely overstated because of the absence of direct measurements, and similar claims with respect to biomass collapse also need careful scrutiny and measurement.

Other work in formulation will focus on the fate of logged areas, both within years and between years. Matricardi (2003) has made considerable progress on this research front, and is also precisely defining the detection limits of the remote sensing methods we currently use. Claims that significant areas of selective logging in Amazonia remain undetected due to logging’s “cryptic” signature and, hence, that significant carbon fluxes are missing in current carbon accounting methods need to be thoroughly tested in future work. Our results to date, however, show that only areas with very low harvest intensity (<10 m$^3$ha$^{-1}$) remain undetectable. The minimal disturbance of these forests therefore also limits their importance as a significant carbon sources.

The focus of the results reported here were, in large part, on advancing the measurement of change into the domain of degradation, to augment past work on deforestation. Although logging is becoming an important forest impact, we conclude that this has only recently been the case, and that previous estimates of deforestation were not significantly underestimating the role of logging. Nonetheless, future work is needed to develop improved measures of degradation through the use of forest fractional cover, techniques which will utilize continuous fields rather than discrete classification of land cover.

Lastly, for a complete picture of change in Amazonia, these observations and measurements will need to be incorporated into models which include the various agents which deforest or degrade the Amazon forested landscape, particularly for prognostic forecasting. There has been very significant progress on this research front already. Cochrane and his colleagues (Laurance et al., 2001; Verissimo et al., 2002) have projected future forest change using a GIS modeling approach to assess the effects of future road infrastructures as well as conservation and development projects. Using a completely different approach, Zhou (2002) has modeled future deforestation rates in an econometric framework that takes into account ranchers, loggers, miners, small holders and many additional demographic and economic factors. Her results from this econometric analysis agree quite closely with the GIS-modeling of Laurance et al (2001), but more work is needed to spatialize the econometric modeling and directly couple it to remote sensing.

With considerable work ahead, it is nonetheless important to reflect on the value of recent efforts within the LCLUC program. Through the use of direct measurements, a solid basis for quantification and analysis of the full suite of forest impacts in Amazonia has been provided. The knowledge of land cover and land use change processes and patterns are enhanced by the increased spatial resolution and growing temporal depth of the available information. Through this progression, our capability to model and predict the complex dynamics of forest change in the Amazon is being brought to life.
10 References


PATTERN TO PROCESS IN THE AMAZON REGION

UNEP. Cochrane, M.A. 2002. Spreading Like Wildfire – Tropical Forest Fires In Latin America And The Caribbean: Prevention, Assessment And Early Warning. United Nations Environment Program, Regional Office for Latin America and the Caribbean. 96pp
TOWARDS AN OPERATIONAL FOREST MONITORING SYSTEM FOR CENTRAL AFRICA

NADINE T. LAPORTE1, TIFFANY S. LIN1, JACQUELINE LEMOIGNE2, DIDIER DEVERS3, MIROSLAV HONZÁK4

1The Woods Hole Research Center, P.O. Box 296, Woods Hole, MA 02543 USA
2NASA Goddard Space Flight Center, Applied Information Sciences Branch, Code 935, Building 28, Greenbelt, MD 20771 USA
3University of Maryland, Department of Geography, College Park, MD 20742 USA
4Conservation International, 1919 M Street, NW, Washington, DC 20036 USA

1 Introduction

Characterizing and mapping land cover and land use change in the rain forests of Central African is a complex process. This complexity is marked by the diversity of land use practices across six different countries (Cameroon, Central African Republic, Democratic Republic of Congo, Equatorial Guinea, Gabon, and Republic of Congo), the lack of full and continuous cloud-free coverage by any single optical remote sensing instrument, and the limited institutional capacity to implement mapping and monitoring activities.

As part of the NASA Land Cover Land Use Change Program (LCLUC) and the Central Africa Regional Program for the Environment (CARPE), an “Integrated Forest Monitoring System” (NASA-INFORMS project; http://www.whrc.org/africa) was established in close collaboration with national forest services, private logging companies, and conservation organizations. This project has been focused on developing remote sensing products for the needs of forest conservation and management, insuring that research findings are incorporated in the forest management plans at the national level.

In this chapter, we will describe how multi-temporal and multi-sensor remote sensing observations and techniques have been integrated with in-situ data for habitat mapping, logging monitoring, and biomass estimation. Using time-series of Landsat imagery, for example, we have mapped the expansion of logging activities in the northern Republic of Congo, providing a new monitoring tool for the national forest service. Indices for estimating the intensity of timber harvesting are also in development. Maps of forest types, as well as deforestation assessment around population centers, have been produced in collaboration with different stakeholders in order to promote better forest management practices in the region.

While these results suggest promise for mapping and monitoring Central Africa’s forests using Landsat imagery, additional data sets and development of new data processing approaches are needed to improve land cover characterization and forest monitoring. We will provide examples of such techniques—one that used a hybrid method combining radar and optical remote sensing to describe vegetation types in Gabon and another that attempted to discriminate different levels of above-ground forest biomass in southern Cameroon using radar data alone.

2 The Central Africa Region

Central Africa contains the world’s second largest block of tropical rain forest (c. 1.8 million km²) after the Amazon (Laporte et al., 1998). While rain forests in West and East Africa already have been reduced to fragments (Brou Yao et al., 2000; Riitters et al., 2000), Central Africa still contains the largest tracks of old-growth forest in the continent (Laporte et al., 1998). The history of deforestation in West and East Africa provides an important lesson for forest conservation and management in Central Africa. Between mid-1970 and mid-1990, total wood production in Ivory Coast had decreased by 50% (Durrieu de Madrone et al., 1998). During the same period, deforestation rates had also increased due to attractive market prices for cacao, unlimited cheap labor from neighboring countries, and a stable political situation that promoted the agricultural sector (Durrieu de Madrone et al., 1998). Today, Ivory Coast’s old-growth rain forests are found only in parks, reserves, and few plantations (Brou Yao et al., 2000); the same situation is true in Ghana and Uganda (Laporte et al., 2002). Under the combined pressure of increased population and favorable economic markets, the future of the Central African rain forest could follow a similar path if the national governments do not adequately plan for the long-term management of their rich forests.

Central Africa is the largest carbon reservoir on the continent (Faure and Faure, 1990). It is also a center of biological diversity, harboring over 400 mammal species, more than 1,000 species of birds, and at least 10,000 plant species, of which about 3,000 are endemic (IUCN, 1992). Recent studies suggest that, after disturbances, species richness and ecosystem resilience of the forest are dependent on the presence and/or abundance of keystone animal species (e.g. Kellman et al., 1996). Many tree species depend on animals for seed dispersal; for example, the role of elephants and primates in seed dispersal is well documented (White 1994a; Wrangham et al., 1994; Hawthorne and Parren, 2000). However, hunting pressure on wildlife populations is strong as Central Africa is one of the poorest regions in the world. Moreover, industrial logging opens up previously remote parts of the forest to human access and thus commercial hunting (Robinson et al., 1999).

One-third of the total population of Central Africa lives in the tropical rain forest. Of the 24 million people, estimated 3 million from 150 different ethnic groups depend strictly on the forest ecosystems for their livelihoods (Bahuchet et al., 2000). These forest hunter-gathers—commonly referred to as pygmies—entertain cultural and commercial exchange with the dominant ethnic groups—diverse populations of Bantu and Ubanguians who traditionally rely on swidden agriculture, fishing, and trapping. Since colonization, most Bantu have been living along the road network, resulting in a relatively high and localized population density (c. 50 inhabitants per km²). The ‘mutual economic dependency’ between the forest hunter-gathers people (providing bushmeat and medicinal products) and the sedentary Bantu (producing crops) is ancient (Joiris and Bahuchet, 1993). However, with the arrival of industrial logging, the hunter-gathers are spending increasingly more time working for the logging companies and/or providing cheap labor for commercial hunting. Coupled with timber extraction, the growing of sedentary population and land use practice around logging towns also increases the overall rate of deforestation in the region.

In Central Africa, farmers clear between 0.5-3.0 ha of forest per family every year, creating a mosaic of degraded vegetation in which new fields are intermingled with older fields of 2-3 years old, fallows, secondary forest, and mature forest. This
complex agro-system mosaic, with an average size of clearing in the order of one hectare, is difficult to monitor using low-resolution satellite imagery. Recent estimates of deforestation using 8-km AVHRR data underestimated the rates of change due to the patchiness of forest loss in this region (Defries et al., 2000). Utilizing 1990 1-km AVHRR satellite images, it was estimated that farmers had converted more than 12% of the original forest in Central Africa (Laporte et al., 1998). Today, the most extensive land use in the region, however, is commercial logging. Currently, 41% of the rain forests in Central Africa have been allocated for timber extraction. While most of the forests in Cameroon and Gabon have been logged at least once (Reitsma, 1988; White, 1994b), the Democratic Republic of Congo and the Republic of Congo still contain extensive amounts of unlogged forests. Table 1 summarizes statistics on population and land use per country. These statistics are, however, uncertain and in need of improved accuracy. In the following sections, we will provide some case studies on the utility of high-resolution imagery, including Landsat optical and JERS-1 radar data, for mapping land cover and monitoring land use change associated with farming and logging.

Table 1. Statistics on the forest, population, and road network of Central Africa.

<table>
<thead>
<tr>
<th>Country</th>
<th>Population (millions)</th>
<th>Population Density (per km²)</th>
<th>Deforestation Rate 1980s-1990s 1990-2000s</th>
<th>Road Density (km/km²)</th>
<th>Forest Cover (%)</th>
<th>Agriculture &amp; Fallow (%)</th>
<th>Logging (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cameroon</td>
<td>15</td>
<td>27.5</td>
<td>0.6, 0.9</td>
<td>5</td>
<td>37</td>
<td>14</td>
<td>31</td>
</tr>
<tr>
<td>Central African Republic (CAR)</td>
<td>3.5</td>
<td>5.2</td>
<td>0.4, 0.1</td>
<td>3</td>
<td>10</td>
<td>10</td>
<td>6</td>
</tr>
<tr>
<td>Dem. Rep. of Congo (DRC)</td>
<td>51</td>
<td>18.3</td>
<td>0.6, 0.4</td>
<td>3</td>
<td>48</td>
<td>4</td>
<td>18</td>
</tr>
<tr>
<td>Equatorial Guinea Gabon</td>
<td>0.35</td>
<td>15.6</td>
<td>0.4, 0.6</td>
<td>11</td>
<td>65</td>
<td>23</td>
<td>61</td>
</tr>
<tr>
<td>Rep. of Congo</td>
<td>1.2</td>
<td>4.9</td>
<td>0.6, n.s</td>
<td>3</td>
<td>80</td>
<td>9</td>
<td>48</td>
</tr>
</tbody>
</table>

(1) From CARPE CD-ROM and based on 1990 population
(2) Annual rate in percent from FAO Tropical Forest Assessment (http://www.fao.org/forestry/fo/fra/index.jsp)
(3) From CARPE CD-ROM (http://carpe.umd.edu)
(4) Derived from AVHRR analyses of the 1990s (Laporte et al., 1998)
(5) From the most recent national forest service databases (changes are underway in Gabon, DRC, and Congo)

3 Habitat Mapping and Land Cover Change Assessment for Forest Conservation and Management: Sangha Tri-National Park Case Study

The needs for forest monitoring are urgent in Central Africa. National institutions in the region lack the most basic information to make land use decision and policy. Recent vegetation maps produced for the region were generated from low-resolution satellite imagery including AVHRR (Laporte et al., 1998), SPOT vegetation (Mayaux et al., 2003), and MODIS (Hansen et al., 2003). These coarse-scale maps are useful for
monitoring land cover at the regional level, but they are poorly adapted to forest management needs at the landscape level (Demaze et al., 2001). In most of Central Africa, changes in land cover associated with agricultural expansion are occurring at such a fine scale that only high-resolution imagery can provide accurate estimates on the extent and the rate of land cover and land use change.

In this section, we will describe how the NASA-INFORMS project produced crucial information for the forest management of northern Republic of Congo. These results include a new land cover map for the Sangha Tri-National Park (section 3.1); the monitoring of logging road progression (section 3.2); and, deforestation rates around major population centers (section 3.3).

Table 2. Time series of Landsat imagery used to monitor logging in northern Republic of Congo from the 1970s to the 2000s. * denotes images used for land cover mapping of the Sangha Tri-National Park.

<table>
<thead>
<tr>
<th>Path-Row</th>
<th>1970s</th>
<th>1980s</th>
<th>1990s</th>
<th>2000s</th>
</tr>
</thead>
<tbody>
<tr>
<td>181-58</td>
<td>n/a</td>
<td>1986-01-16</td>
<td>n/a</td>
<td>2000-03-03</td>
</tr>
<tr>
<td>181-59</td>
<td>n/a</td>
<td>1984-09-07</td>
<td>1999-11-12</td>
<td>n/a</td>
</tr>
<tr>
<td>182-58</td>
<td>1979-03-18</td>
<td>1986-12-09</td>
<td>2002-01-20</td>
<td>*2001-02-09</td>
</tr>
</tbody>
</table>

For this case study, we used fourteen images of the Landsat Thematic Mapper (Table 2). These images were geographically co-referenced to the GEOCOVER products, which are orthorectified Landsat scenes produced by the Earth Satellite Corporation (EarthSat). Ground control points collected at road intersections or other land features were used to assess the accuracy of geo-referencing. The highest locational error was in the order of 1-pixel (c. 30 meters).

3.1 FOREST HABITAT MAPPING

Biodiversity conservation depends strongly on the management of both protected areas and their buffer zones. Unfortunately, in most of Central Africa, very little is known about the distribution of vegetation types and associated threats, such as deforestation, forest degradation, and forest fragmentation. While forest covers roughly 45% of Central Africa, only 10%, or 180,000 km², is currently protected (Laporte et al., 1998). Moreover, the park system is very fragmented. The viability of wildlife populations is not likely to be sustainable if the parks are connected by corridors (e.g. Oates, 1996). As part of the USAID-Central Africa Regional Program for the Environment (CARPE), a new landscape approach has been adopted to reduce forest loss by increasing natural resource management capacity and by creating corridors between parks.

In collaboration with CARPE partners (Wildlife Conservation Society, “Projet de Gestion des Ecosystèmes Périphériques au Parc National Nouabalé-Ndoki), a new vegetation map was produced for the Sangha Tri-National Park. Given the presence of heavy cloud cover and atmospheric haze in most of the Landsat scenes, various combinations of spectral bands were used to generate land cover maps from “semi-unsupervised” classification. Although the two Landsat scenes were classified
independently, the general procedures remained the same. The first step consisted of
masking of clouds, cloud shadow, and water using mainly band 3, 4, and 5 and an
ISOCLUS algorithm. The second step consisted of running unsupervised classification
on the cloud-free data using various combinations of bands 2, 3, 4, and 5. Several
iterations were performed to separate different forest types, and the clustering results
were aggregated into 18 classes based on spectral separability and contextual
information. Then, the aggregated image was filtered using a 4-pixel sieve filter, where
clusters of less than 4-pixels were merged with the largest nearest neighboring cluster.
Finally, each cluster was assigned to a thematic class based on a pre-defined legend
established with end-users (park managers and forest service). In some cases, manual
editing was performed when two habitats important for wildlife management could not
be differentiated based on their spectral signatures.

The resulting map, with 18 different land cover & land use categories, is a
significant step towards better characterization of wildlife habitats in the area.
Distribution, structure, and phenology of vegetation are the most important
determinants of wildlife population distribution and density (White, 1994a). This map
is thus useful for modeling the distribution of wildlife populations. It also provides
important baseline information for comparing the extents of different habitats across the
Sangha Tri-National Park (Table 3). For example, while the semi-evergreen mixed
species forest is the dominant habitat in all three sections of the park complex, the
evergreen monodominant *Gilbertiodendron dewevrei* forest occurs in large patches
mainly in the Nouabalé-Ndoki National Park. The distribution and extent of this prime
seasonal foraging habitat for wildlife lays on a south-to-north gradient, with the highest
percent cover and the largest stands found in the Nouabalé-Ndoki National Park
(Republic of Congo), the Loundoungou concession, and the Lobéké Reserve
(Cameroon). Conversely, savanna habitat is found only in the Dzanga-Sangha Special
Reserve (Central African Republic).

<table>
<thead>
<tr>
<th>Land Cover Type</th>
<th>Nouabalé-Ndoki</th>
<th>Dzanga-Sangha</th>
<th>Lac Lobéké</th>
</tr>
</thead>
<tbody>
<tr>
<td>Semi-evergreen mixed species forest</td>
<td>73</td>
<td>89</td>
<td>85</td>
</tr>
<tr>
<td>Evergreen <em>Gilbertiodendron dewevrei</em> forest</td>
<td>23</td>
<td>5</td>
<td>9</td>
</tr>
<tr>
<td>Swamp and marsh</td>
<td>3</td>
<td>3</td>
<td>5</td>
</tr>
<tr>
<td>Savanna (include forest-savanna transition)</td>
<td>0</td>
<td>2</td>
<td>0</td>
</tr>
<tr>
<td>Others</td>
<td>1</td>
<td>1</td>
<td>1</td>
</tr>
</tbody>
</table>

### 3.2 LOGGING MONITORING

Until recently, the Sangha Tri-National Park was one of the most remote places on
Earth, with a large population of Aka and Baka forest people (Moukassa, 2001). Since
1970s, industrial logging has been attracting migrant workers from across Central
Africa, resulting in demographic change and population booms in logging towns. Nine
logging companies are active in northern Congo, and an extensive network of logging
roads now surrounds the Sangha Tri-National Park (Figure 1). Today, this increase of
access into the intact forests is recognized as one of the most critical factors leading to
large-scale biodiversity loss.
Monitoring the expansion of logging roads is an important part of wildlife conservation in Central Africa. Road access into remote forest is often followed by an increase in commercial hunting that, if not properly managed and controlled, can threaten wildlife populations (Robinson et al., 1999). Using a time-series of Landsat satellite imagery (Table 2), roads were digitized to monitor the progression of logging in northern Congo and to identify areas potentially threatened by poaching. Currently, new roads are being built at a rate of more than 100 km per year per logging company. Between 2001 and 2002, the road network had increased by 176 km in the Lopola concession and 165 km in the Mokabi concession. We also tracked the total extent of roads constructed since the beginning of industrial logging in the 1970s. To date, this amounts to more than 5,000 km of logging roads—twice the total length of the primary roads in the entire country (Laporte and Lin, 2003).

The intensity of logging, in terms of number of trees harvested per unit area, was also estimated using Landsat imagery. Number of trees removed per 50-ha harvesting parcel was obtained from CIB (Congolaise Industrielle du Bois) for a 9,000 ha area in the Kabo concession, logged between 1998 and 1999. For each parcel, the total number of trees harvested was correlated with the proportion of various land cover types mapped using a dry season Landsat ETM+ imagery of 2001. (Refer to Section 3.1 for details on the vegetation mapping.) Of all land cover types, we found that logging intensity was most strongly correlated with the total amount of exploitable
forest, with “exploitable” being defined by the logging companies as the extent of *terra firma* semi-evergreen mixed species forest (Figure 2).

![Figure 2. Estimating logging intensity with Landsat imagery two years post-harvesting (n = 143; \( r^2 = 0.59; p < 0.01 \)).](image)

### 3.3 MONITORING DEFORESTATION

Northern Congo has been undergoing various degrees of deforestation. Little is known on the current rate and extent of land cover conversion from forests to agro-systems. In this case study, we focused on comparing deforestation in the last decade around four major population centers—Ouesso, Pokola, Kabo, and Ngomb—in the vicinity of the Nouabalé-Ndoki National Park. Ouesso, with a population of approximately 18,000, is an old administrative town and a center of bushmeat trade (Thuret, 1997). Pokola, Kabo, and Ngombe are logging towns located along the Sangha River, which remains the main communication network between settlements. Until the 1970s, Pokola was a small fishing village with only a few hundred inhabitants. It is now the headquarters of CIB, one of the largest industrial logging companies in the region. With a logging-driven economy, the population in Pokola has increased to over 7,000 (Moukassa, 2001). Managing more than 1.5 million hectares of forests, CIB also operates from Kabo, a town of c. 1,400 (Moukassa, 2001). Ngombe, the operation center for the Danzer logging company, has an unknown population size but is considerably smaller than Pokola.

Between 1990 and 2001, the greatest forest loss occurred around the two larger towns of Ouesso and Pokola. However, marked differences exist in the pattern of deforestation between the two. In 1990, Ouesso was already surrounded by a large area of young secondary forest (c. 30 km\(^2\)), which probably had been cleared previously or was highly degraded. This “degraded” or “regrowth” forest is typically preferred for shifting cultivation or housing conversion, consequently reduces the pressure of creating new forest clearings. Pokola, on the other hand, had limited area of young secondary forest. Most of the conversion to farmland and “urban” land cover in the last decade was, therefore, from old secondary forest that had only been selectively logged in the past. New forest clearings were created as the amount of 15-20 year-old young
secondary forests was limited around Pokola. Furthermore, note that, in Ouesso, the area of increase in agriculture almost equals that in bare soil, but the increase in farmland in Pokola is only a third of the increase in “urban” build-up. This observation corresponds to the fact that more people in Ouesso rely on farming for their livelihood than in Pokola, of which nearly all inhabitants are employed by CIB. The spatial pattern of land cover also differs between the administrative town and the logging towns, with more “diffuse” land cover conversion in Ouesso and more “compact” conversion in the immediate vicinity of Pokola, Ngombe, and Kabo.

4 Testing New Approaches for Land Cover Mapping and Biomass Estimation

Central Africa, similar to many tropical regions, lacks full and continuous cloud-free coverage by any single high-resolution optical remote sensing instrument (i.e. Landsat). Therefore, it is crucial to develop multi-sensor hybrid approaches to characterize land cover types and to monitor human pressure on the forests. In the following sections, we will describe pilot studies of habitat mapping using radar and optical data combined (section 4.1) as well as biomass estimation using radar imagery (section 4.2).

4.1 FUSION OF SAR AND OPTICAL DATA FOR LAND COVER MAPPING OF THE LOPE RESERVE

Figure 3 illustrates the idea where the fusion of high-resolution radar data and low-resolution optical data is able to yield land cover maps of equivalent or better quality than those created by high-resolution optical data alone. Such an approach, if successful, could compensate for the lack of good Landsat coverage in the coastal region of Central Africa where heavy cloud cover traditionally limits the use of high-resolution optical data. While our future work will involve comparing land cover classification using JERS and MODIS data combined with that using Landsat data alone, we tested this concept of data fusion using 6-m SAR data and 30-m Landsat imagery in the pilot study.

The objective was to map eight different vegetation types for the Lopé Reserve in Gabon: 1) Montane Forest; 2) Mixed Forest—pluri-stratum stand with at least 3 different tree heights, including riparian and Marantaceae forest; 3) Okoumé Forest—mono-stratum stand with trees of similar diameters and heights; 4) Savanna in the range of 1 to 1.5 meters in height; 5) Fern Savanna of less than 1 meter in height; and, 6) Burnt Savanna. The higher resolution data (6-m) was extracted from a SAR image acquired from the “Mission Aéroportée radar SAR” flown in 1994; the lower resolution data (30-m) was a subset of the Landsat TM imagery of Path 185-Row 60, acquired on May 1, 1988. The Landsat data set was resampled to 6-m resolution since it was used as the reference image and as part of the fusion.
Multi-sensor data fusion may be performed either before the classification step with a pixel-level image fusion technique, or as a post-processing by combining multiple classification results at the feature- or decision-level (Pohl et al., 1998). In this experiment, we performed the fusion before the classification and used a number of hybrid fusion techniques. The results described here are based only on the use of a wavelet transform, which describes images in the frequency domain at multiple spatial resolutions using Multi-Resolution Analysis (MRA). Figure 4 illustrates the wavelet-based fusion idea, where high- and low-resolution data are independently decomposed using the MRA scheme. Briefly, a wavelet decomposition of any given signal (1-D or 2-D) is the process which provides a complete representation of the signal according to a well-chosen division of the time-frequency (1-D) or space-frequency (2-D) plane. Through iterative filtering using low- and high-pass filters, it provides information about low- and high-frequencies of the signal at successive spatial scales. Performing image fusion within a wavelet framework enables the user to fuse data selectively at various frequency components in the lower spatial resolutions while preserving spectral information of higher spatial resolutions. During the reconstruction phase, components from both decompositions are combined to create the new fused data. In our experiments, we used a Daubechies filter size 4 (Daubechies, 1991) and a Mallat MRA (Mallat, 1989) for both decomposition and reconstruction and for both types of data. Low-frequency information of the lowest spatial resolution data (i.e., Landsat) were combined with high-frequency information of the highest resolution data (i.e., SAR) in order to take simultaneous advantage of the higher spatial and spectral resolutions. However, one can imagine using different filters for decomposition and reconstruction as well as for high- and low-resolution data that would better preserve the spatial, spectral and textural properties of the data.
Due to the lack of reliable training data sets at the time of the experiment, unsupervised classification was used to obtain a vegetation map from the Landsat data alone and from the fused data. With the targeted number of clusters set to 10 and all other parameters being equal, unsupervised classification using the ISOCLUS algorithm returned 9 clusters for both data sets. These clusters then were regrouped thematically into the six vegetation types described earlier. Although qualitatively similar, the fused clustering shows more localized details with differentiation of the three savannas types and a different clustering of the Montane Forest versus Mixed Forest. This pilot study, although incomplete, illustrates that a wavelet-based fusion of radar and optical data can augment a forest monitoring system for applications at the local and regional scales. Our ongoing work involves data fusion of JERS and MODIS imagery compared to Landsat and to MODIS data alone; investigation of different types of wavelet filters; improvements to the clustering scheme; and, quantitative evaluation of results using in-situ data.

4.2 ASSESSING FOREST BIOMASS USING JERS-1 RADAR MOSAICS

Studies on changes in above-ground biomass under different land use scenarios have been focused primarily on the Amazon (e.g. Fearnside and Guimaraes, 1996). In Central Africa, however, very little is known about the impacts of agricultural conversion and logging on carbon stocks or cycling. Many studies have shown positive correlation of radar backscatter to total above-ground biomass in different forest types in the Northern Hemisphere (e.g. LeToan et al., 1992; Ranson and Sun, 1994). Radar backscatter from forest canopies depends on wavelength, incidence angle and polarization, as well as canopy structure and wetness. Generally, longer wavelength...
(L-band) SAR imagery such as JERS-1 may be used to discriminate between different levels of forest biomass up to a certain saturation point. Additionally, there is evidence that cross-polarized backscatter is more sensitive to changes in biomass (Tiangco and Forester, 2000). The potential for mapping forest biomass using SAR data is, however, limited when the forest structure is complex and the biomass level is high, as in the case of most tropical forests in Central Africa.

Figures 5a, 5b. Estimating above-ground biomass with JERS-1 normalized backscatter: a) February 1996, dry season ($r^2 = 0.15; p < 0.05$); b) November 1996, wet season ($r^2 < 0.01; p > 0.05$).

We explored the utility of 100-m JERS-1 radar imagery for above-ground biomass estimation across a range of 1-ha plots in 61 study sites throughout southern Cameroon. The field biomass measurements, collected in 1995, were compared with the normalized backscatter of the 1996 JERS-1 mosaic produced by the NASA Jet Propulsion Laboratory (http://trfic.jpl.nasa.gov). We found poor relationships between the above-ground biomass measurements and backscatter for both low and high water mosaics (Figures 5a & 5b). These findings suggest limited utility of JERS-1 radar imagery for biomass estimation across tropical Africa. Similar limitations have been noted in the Amazonian forests (Salas et al., 2002). There are, however, several possible factors for the poor results. Locational errors of the field sites and geographic
mis-registration of the images could negatively affect the backscatter-biomass relationship. The pre-processing of the JERS-1 mosaic could also have been a factor, particularly owing to resolution loss due to the resampling of the data. Moreover, forest structure in these areas is highly variable, especially in disturbed sites. Comparison of 1-ha plots with the 100-m JERS-1 mosaic therefore could have been negatively influenced by the high spatial heterogeneity of above-ground biomass. However, we believe that the site heterogeneity was adequately characterized since the sampling method was based on that of the Cameroonian forest service designed for biomass estimation and has been proven accurate for related studies using 1-km AVHRR data (Boyd et al., 1999; Lucas et al., 2000). Finally, JERS-1 normalized backscatter is likely to underestimate biomass of disturbed forests where a collection of numerous regenerating small trees may have the same level of biomass as a mature stand with few big trees. This limitation was also noted in boreal forests (Ranson and Sun, 1994). In the future, we plan to explore the use of LIDAR waveform models, which should allow improved estimates of both forest biomass and canopy structure.

5 Conclusions

International efforts are underway to improve operational forest monitoring. At the first Central Africa “Global Observation of Forest Cover” workshop held in February 2000 in Libreville, Gabon, several issues limiting the development of operational forest monitoring systems were identified by national forest services and their international partners. These limitations included lack of technical and financial resources, poor access to data and information (including the internet), and lack of training facilities and opportunities. It was also unanimously recognized that remote sensing is a key component to any forest monitoring system and that, at each stage of scientific understanding, there be transfer of relevant information to policy and decision makers.

While information derived from remote sensing products can help natural resource managers to determine optimal management practices for maintaining healthy forest ecosystems, few tropical countries have the resources to develop their own remote sensing programs. Initiatives from the NASA/LCLUC program, the CARPE program, and GOFC can, therefore, help to facilitate the transfer of technology to the national forest services in Central Africa. Ultimately, these activities must contribute to the better management of natural resources for the present and future generations by the residents of the region. Our development of an integrated forest monitoring system, where remote sensing technology is combined with in situ data from commercial logging and biodiversity inventories, allowed us to provide crucial information for the management of the northern Congo forest ecosystem and to explore the utility of new approaches to land cover mapping. We continue to pursue these activities in a region that is one of the least known yet most productive and biologically diverse on earth.

Acknowledgments. This research was supported by the NASA Land Cover Land Use Change Program and the Central Africa Regional Program for the Environment (CARPE). It would not have been possible without the logistic support of the Wildlife Conservation Society (Paul Elkan and Fiona Maisels) and the Congolaise Industrielle des Bois (Olivier Desmet, Dominique Paget). We want to thank the following individuals for generous donation of their time and their work dedicated to preserving these unique ecosystems: Steve Blake, Brian Curran, Bourges Djoni-
Djimbi, Sarah Elkan, Mike Fay, Steve Gulick, Richard Malonga, Antoine Moukassa, and Chris Wilks. We also thank Yves Dubois, Leon Embon, Patrick Geffroy, Frederic Glannaz, Jackie Glannaz, Patrice Gouala, Christian Guyonvaro, Gregoire Kossa, Tom van Loon, Alfred Tira, and Luca van Der Walt for field assistance and working towards the sustainable use of the region’s forest resources. Scott Goetz provided editorial assistance; Debra Fischman and Jeremy Goetz assisted with compiling the road network.

6 References


CHAPTER 7

LAND USE AND LAND COVER CHANGE IN SOUTHEAST ASIA

JAY H. SAMEK¹, DO XUAN LAN², CHAOWALIT SILAPATHONG³, CHARLIE NAVANAGRUHA⁴, SHARIFAH MASTURAH SYED ABDULLAH⁵, IWAN GUNAWAN⁶, BOBBY CRISOSTOMO⁷, FLAVIANA HILARIO⁸, HOANG MINH HIEN⁹, DAVID L. SKOLE¹, WALTER CHOMENTOWSKI¹, WILLIAM A. SALAS¹⁰, HARTANTO SANJAYA¹¹

¹Center for Global Change and Earth Observations, Michigan State University, East Lansing, MI, USA
²Forest Inventory and Planning Institute, Ministry of Agricultural and Rural Development, Hanoi, Vietnam
³Geo-Informatics And Space Technology Development Agency, Bangkok, Thailand
⁴Mahidol University, Bangkok, Thailand
⁵Earth Observation Centre, Universiti Kebansaan Malaysia, Selangor, Malaysia
⁶Bureau of Programme Coordination and External Relations, ASEAN Secretariat, Jakarta Indonesia
⁷Database Management Division, National Mapping and Resource Information Authority, Manila, Philippines
⁸Climatology and Agrometeorology Branch, Philippine Atmospheric, Geophysical and Astronomical Services Administration, Quezon City, Philippines
⁹Disaster Management Center, Standing Office of the Central Committee for Flood and Storm Control, Hanoi, Vietnam
¹⁰Applied Geosolutions, LLC, Durham, NH, USA
¹¹BPPT (Agency for Assessment and Application of Technology), Jakarta, Indonesia

1 Introduction

Southeast Asia is a culturally, environmentally, and geographically rich, diverse, and dynamic region. Comprised of eleven countries, it spans the Indochina and Malay peninsulas and the Malay Archipelago. Five nations, Cambodia, Laos, Myanmar, Thailand, and Vietnam, are entirely on the mainland. The remaining six, Brunei, East Timor, Indonesia, Malaysia, Philippines, and Singapore, are spread across thousands of islands. Coastal zones and river deltas, piedmont zones and mountain chains, with peaks reaching heights greater than 19,000 feet¹, characterize the region. The land cover and land use change patterns evident in Southeast Asia are as diverse and dynamic as the political, economic, and demographic spheres in these eleven nations.

Direct observations and measurements derived from satellite data, particularly the MSS, TM and ETM+ sensors aboard the suite Landsat spacecrafts dating back to the early 1970s, provide empirical data used to document the spatial extent of land cover and land use, and also their rates of change over time. The region has experienced loss of forest cover as a whole, particularly hardwood *dipterocarp* species and

¹ Hkakabo in northern Myanmar is the highest peak in Southeast Asia at 19,296 feet above sea level.
mangrove forests, due to a variety of proximate and distant drivers. Recent research, however, provides evidence of dynamic forest re-growth and clearing cycles in upland swidden systems (Skole et al., 2002; Wang 2003). Conservation and resource protection efficacy have also reversed the trend of forest clearing in some regions. In some cases, the total area in forest cover has changed little over time, but the forest cover itself has been converted to plantation forests, such as rubber and oil palm, in place of more diverse, species-rich natural forest. Furthermore, as in other regions of the tropics, forest degradation is rapidly becoming as much of a threat to forest cover as outright clearing and conversion (Achard et al., 1998; Matricardi et al., 2001; FAO 2001; UNEP 2002).

Figure 1. Indochina Forest Cover – 1999 (ETM+). Dark gray – forest, black – water. (See CD for color image.)

Accurate measurements of the forest cover and of forest cover changes in Southeast Asia are important for understanding global and regional climate change, the impacts on biodiversity, ecological health, and human welfare. An equally important dimension of land cover and land use in Southeast Asia is that of urban growth. With
LAND USE AND LAND COVER CHANGE IN SOUTHEAST ASIA

Cities that are large in terms of population and area\(^2\) such as Manila, Bangkok, Jakarta, Hanoi, and Kuala Lumpur, Southeast Asia’s urban environment growth is of global importance socially and with respect to land use and the resultant urban impact on land cover.

2 Regional Forest Cover

From a long-term perspective, 300 years, regional forest cover in Southeast Asia has experienced a steady decline (Grubler 1994; Fu et al., 1998). Decreases in forest cover since the 1700s coincide with the expansion of agriculture, the region’s integration into a global economy, and population growth. Rapid economic development in the region, and technological advances since the mid 1900s, have contributed significantly to the loss of forest cover. The development of transportation networks providing access to areas traditionally traversed on foot, the invention of high-tech machinery to log forests and extract minerals, the construction of hydropower facilities to support the energy demands of the growing urban populations and to support national efforts to compete in the global market economy, the development of large-scale cash-based agriculture in place of subsistent farming, industrial development, national policies favoring economic development over environmental conservation and weak political control have all contributed to the decline in forest cover in Southeast Asia.

Long-term estimates of forest cover loss, agricultural expansion, and population growth for Asia as a whole show a loss of $108 \times 10^6$ hectares of forest, an increase of $128 \times 10^6$ hectares of cropland, and a population increase of $582 \times 10^6$ people from 1700 – 1920. Between 1920 and 1980 Asia experienced an additional loss of $142 \times 10^6$ hectares of forest, an additional increase of $185 \times 10^6$ hectares of cropland, and an additional population increase of $1562 \times 10^6$ people (revised from Fu et al., 1998). To conclude that population and agricultural expansion alone drive deforestation is too simplistic. However, these proximate causes certainly contribute heavily to the loss of forest cover.

The global acquisition of remotely sensed satellite data, since the early 1970s, has contributed significantly to our basic understanding of the forest cover dynamics in this region. The NASA Landsat Pathfinder Humid Tropical Forest Project (HTFP) has measured the extent of forest cover in Indochina at four time periods using high-resolution Landsat MSS (1973, 1985), TM (1992) and ETM+ (1999) data. The Pathfinder HTFP analyses are the only empirically derived, wall-to-wall measurements of Indochina at 30-meter (TM and ETM+) and 60-meter (MSS) spatial resolutions. The Pathfinder HTFP results for Southeast Asia show a decline in forest cover from $114.72 \times 10^6$ hectares in 1973 to $92.87 \times 10^6$ hectares in 1999, an overall loss of $21.85 \times 10^6$ hectares at an average annual rate of $0.84 \times 10^6$ hectares over the twenty six year time frame (revised from Skole et al., 1998 and MSU unpublished results 2003). Figure 2 shows, however, that the annual rate of deforestation in Indochina peaked in the 1980s and has declined since. The annual forest cover loss between 1973 and 1985 was $1.31 \times 10^6$ hectares, whereas between 1985-1992 and 1992-1999 the rates were 0.50 and 0.37 x 10^6 hectares respectively. In relative terms to the total land area of Indochina

(190.036 x 10^6 ha), forest cover has declined from 60.36 % of the total land area in 1973 to 52.08 % in 1985, 50.24 % in 1992 and 48.87 % in 1999. By comparison, the Food and Agricultural Organization of the United Nations’ Forest Resource Assessment 2000 reports Indochina forest cover at 42.57 % of the total land area. The FAO-FRA 2000 calculation is based on country reported statistics that range in years from 1989 in the case of Laos to 1998 for Thailand.

![Figure 2. Indochina Annual Rate of Forest Cover Decline](image)

3 Dynamics of Forest Cover Change

While synoptic analyses, such as the Pathfinder HTFP and FAO FRA studies, show continuous decline in regional forest cover for Indochina and Southeast Asia as a whole, studies at the local scale reveal a more dynamic sequence of land cover and land use change and a non-linear pattern to forest re-growth and clearing. Inter-annual analysis of Landsat data for the Mae Chaem watershed in Chiang Mai, Thailand indicates swidden farming systems in the uplands of northwest Thailand allow for secondary forest regrowth (Silapathong et al., 2002; Wang 2003). Tamdoa National Park in northern Vietnam has seen an increase of forest cover since the park’s establishment (Lan et al., 2002). Forest re-growth has also been documented in the northern part of Stoeung Trang district of Kampong Cham province, Cambodia and is believed to be linked to the decline of the Khmer Rouge regime (Vina et al., 2002).

The upland swidden/fallow areas of Southeast Asia, when analyzed at fine temporal scales, indicate a dynamic system of land cover and land use change that is not captured through decadal analyses. Such changes have been documented in northern Thailand (Skole, et al., 1998; Silapathong et al., 2002). The Mae Chaem district, in Chiang Mai, Thailand is characterized by evergreen hill forest, mixed
deciduous forest, dry *dipterocarp* forest, forest plantation, shifting cultivation, paddy field, field crop, mixed orchards, open land and settlement areas. Mae Chaem is a mountainous area with an elevation range of 300 - 1,800 meters. The analysis of annual Landsat data for a ten-year period (1990 – 1999) for Mae Chaem shows the inter-annual variation of forest clearing and re-growth. While there is an overall decline in forest cover from 1990 to 1999, in two of the ten-year annual increments, 1991-1992 and 1993-1994 the total area of forest increased. Table 1 shows the comparison of land cover and land use changes in Mae Chaem at the annual increment of analysis. Even more significant are the results of the change detection analysis. As indicated in Table 2, and consistent with the synoptic, regional analysis, there is conversion of forest cover to agriculture in nearly all years. However, there are also areas that are converting back to forest from agriculture. These changes support the hypothesis that deforestation in upland swidden regions of Southeast Asia is not a linear cycle of forest clearing and permanent conversion to non-forest land cover but rather, cycles of clearing and abandonment that allow for periods of forest re-growth. The study by Skole et al. (1998) based on a Landsat dataset over a five-year period from 1989 – 1994 also supports these conclusions.

Table 1: Annual Increment of Land Cover and Land Use Change, Mae Chaem, Thailand in Square Kilometers (km$^2$).

<table>
<thead>
<tr>
<th>Class</th>
<th>90-91</th>
<th>91-92</th>
<th>92-93</th>
<th>93-94</th>
<th>94-95</th>
<th>95-96</th>
<th>96-97</th>
<th>97-98</th>
<th>98-99</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest</td>
<td>-2.917</td>
<td>0.936</td>
<td>-7.826</td>
<td>2.289</td>
<td>-0.602</td>
<td>-2.603</td>
<td>-1.318</td>
<td>-1.473</td>
<td>-0.737</td>
</tr>
<tr>
<td>Agriculture</td>
<td>3.106</td>
<td>-1.167</td>
<td>11.366</td>
<td>-6.782</td>
<td>4.871</td>
<td>3.674</td>
<td>2.699</td>
<td>2.322</td>
<td>0.491</td>
</tr>
<tr>
<td>Open land</td>
<td>-0.212</td>
<td>0.172</td>
<td>-3.826</td>
<td>4.424</td>
<td>-4.269</td>
<td>-1.107</td>
<td>-1.517</td>
<td>-0.937</td>
<td>0.324</td>
</tr>
<tr>
<td>Urban</td>
<td>0.033</td>
<td>0.049</td>
<td>0.085</td>
<td>0.078</td>
<td>0.078</td>
<td>0.007</td>
<td>0.102</td>
<td>0.069</td>
<td>0.029</td>
</tr>
<tr>
<td>Water</td>
<td>-0.010</td>
<td>0.010</td>
<td>0.001</td>
<td>-0.009</td>
<td>0.000</td>
<td>0.029</td>
<td>0.034</td>
<td>0.019</td>
<td>-0.107</td>
</tr>
</tbody>
</table>

In addition to upland dynamics of forest clearing and abandonment to secondary growth, there is also evidence from satellite analyses showing forest re-growth in protected areas. Efficacy of protected areas is not uniform across the region or, in some cases, even within any particular nation’s borders. However, the case of forest cover change in Tamdao National Park is exemplary of how the enforcement of natural resource protection policies can foster the expansion of forest cover in a particular area.

Tamdao National Park is an elongated area, oriented in a northeast/southwest direction, which spans three provinces: Vinh Phuc, Thai Nguyen and Tuyen Quang. It comprises a mountain range more than 80 kilometers long, the center of which is located about 80 kilometers northwest of Hanoi, the capital of Vietnam. Tamdao is a tail of arch shaped mountain ranges located in the upper Chay River area. The tails of these ranges converge in Tamdao, and their heads spread out as a fan to the north. The Tamdao range consists of two dozen peaks linked with each other to form sharp edges. The peaks have an elevation around 1000m. The highest peak is Tamdao North (1592m) and is located in the center of the range at a cross point of the three provincial borders. The width of the Tamdao range varies from between 10 and 15 kilometers and is characterized by very steep sloping sides (averaging 26 – 35 degrees). Forests in Tamdao can be classified under tropical forest with a predominance of *Dipterocarpaceae* and *Lauraceae* species. In Tamdao National Park there are six forest
types classified by the Vietnamese Forestry and Inventory Planning Institute (FIPI): evergreen tropical rain forest, evergreen sub-tropical rain forest, scrub forest at the top of mountains, secondary forest after over-logging, regenerating forest, and forest plantation.

Table 2: Percent of Annual Increment of Land Cover and Land Use Change, Mae Chaem, Thailand (km$^2$)

<table>
<thead>
<tr>
<th>Class</th>
<th>90-91</th>
<th>91-92</th>
<th>92-93</th>
<th>93-94</th>
<th>94-95</th>
<th>95-96</th>
<th>96-97</th>
<th>97-98</th>
<th>98-99</th>
</tr>
</thead>
<tbody>
<tr>
<td>Unchanged Forest</td>
<td>94.33</td>
<td>94.28</td>
<td>93.21</td>
<td>93.30</td>
<td>93.52</td>
<td>93.18</td>
<td>93.01</td>
<td>92.7</td>
<td>92.71</td>
</tr>
<tr>
<td>Forest to Agriculture</td>
<td>0.37</td>
<td>0</td>
<td>0.96</td>
<td>0.12</td>
<td>0.18</td>
<td>0.43</td>
<td>0.19</td>
<td>0.34</td>
<td>0.19</td>
</tr>
<tr>
<td>Forest to Open land</td>
<td>0</td>
<td>0.05</td>
<td>0.25</td>
<td>0.03</td>
<td>0.04</td>
<td>0.05</td>
<td>0.05</td>
<td>0.06</td>
<td>0.04</td>
</tr>
<tr>
<td>Unchanged Agriculture</td>
<td>3.72</td>
<td>3.98</td>
<td>3.74</td>
<td>4.55</td>
<td>4.55</td>
<td>5.25</td>
<td>5.68</td>
<td>6.06</td>
<td>6.23</td>
</tr>
<tr>
<td>Agriculture to Forest</td>
<td>0</td>
<td>0.16</td>
<td>0.19</td>
<td>0.41</td>
<td>0.12</td>
<td>0.04</td>
<td>0.05</td>
<td>0.11</td>
<td>0.10</td>
</tr>
<tr>
<td>Agriculture to Open land</td>
<td>0.05</td>
<td>0.06</td>
<td>0.09</td>
<td>0.61</td>
<td>0.01</td>
<td>0.04</td>
<td>0.09</td>
<td>0</td>
<td>0.11</td>
</tr>
<tr>
<td>Unchanged Open land</td>
<td>1.27</td>
<td>1.26</td>
<td>0.45</td>
<td>0.76</td>
<td>0.80</td>
<td>0.65</td>
<td>0.39</td>
<td>0.35</td>
<td>0.23</td>
</tr>
<tr>
<td>Open land to Forest</td>
<td>0</td>
<td>0</td>
<td>0.05</td>
<td>0.03</td>
<td>0</td>
<td>0.05</td>
<td>0.04</td>
<td>0.13</td>
<td>0.04</td>
</tr>
<tr>
<td>Open land to Agricult.</td>
<td>0.12</td>
<td>0.06</td>
<td>0.86</td>
<td>0.01</td>
<td>0.59</td>
<td>0.15</td>
<td>0.31</td>
<td>0.05</td>
<td>0.13</td>
</tr>
</tbody>
</table>

Lan et al. (2002) analyzed three Landsat scenes in a study of land cover and land use change for Tamdao National Park. The study classified data from 1975 (MSS), 1992 (TM), and 1999 (ETM+). The results of the analysis are shown in table 3. Between 1975 and 1992, the area of old growth forest and secondary forests declined 19.7 and 4.3 percent respectively. However, between 1992 and 1999, the area of secondary forest cover expanded by 98.5 percent, or 6,665.4 hectares, nearly doubling the area observed in 1972. Unfortunately, the area of old growth forest declined 22.9 percent for this same time period, a change of 4,515.6 hectares. However, 4,895.7 hectares, the vast majority of transitioned old growth forest in 1992 converted to secondary growth in 1999, rather than to other land cover or land use types (Table 4).

Episodic political events, particularly in Southeast Asia, a region characterized by a varied and diverse social, economic and political past history and present make-up, can also contribute to the patterns of land cover and land use measured using satellite data. Political economy and political ecology studies are emphasizing the importance of social and economic factors that influence land cover and land use patterns beyond population growth and agricultural expansion (Blaikie and Brookfleid, 1987; Hecht and Cockburn, 1990; Arizpe et al., 1995). Vina et al. (2002) document the relationship between the collapse of the Khmer Rouge regime and forest cover expansion in a part of Kampong Cham Province, using a two-date study of land cover and land use change with Landsat data from 1984 (MSS) and from 1991 (TM) and geographic information system modeling (GIS).
Table 3: Multi-date analysis of land cover in hectares for Tamdoa National Park

<table>
<thead>
<tr>
<th>Land use/land cover</th>
<th>1975</th>
<th>1992</th>
<th>1999</th>
</tr>
</thead>
<tbody>
<tr>
<td>Old forest</td>
<td>24,601.0</td>
<td>19,746.7</td>
<td>15,231.1</td>
</tr>
<tr>
<td>Secondary forest</td>
<td>7,073.1</td>
<td>6,769.9</td>
<td>13,435.3</td>
</tr>
<tr>
<td>Open/bare land</td>
<td>9,827.2</td>
<td>13,554.9</td>
<td>12,826.5</td>
</tr>
<tr>
<td>Agriculture</td>
<td>1,624.1</td>
<td>3,053.9</td>
<td>1,632.5</td>
</tr>
<tr>
<td>Total</td>
<td>43,125.4</td>
<td>43,125.4</td>
<td>43,125.4</td>
</tr>
</tbody>
</table>

Table 4: Confusion matrix from 1992-1999 analysis of land cover in hectares for Tamdao National Park

<table>
<thead>
<tr>
<th></th>
<th>99</th>
<th>92</th>
<th>Land uses</th>
</tr>
</thead>
<tbody>
<tr>
<td>OF</td>
<td>14,191.3</td>
<td>4,895.7</td>
<td>Old Forest</td>
</tr>
<tr>
<td>SF</td>
<td>434.1</td>
<td>3,359.6</td>
<td>Second. For.</td>
</tr>
<tr>
<td>OP</td>
<td>597.0</td>
<td>4,988.6</td>
<td>Open Land</td>
</tr>
<tr>
<td>AG</td>
<td>8.7</td>
<td>191.4</td>
<td>Ag. Land</td>
</tr>
<tr>
<td>Total</td>
<td>15,231.1</td>
<td>13,435.3</td>
<td>Total</td>
</tr>
</tbody>
</table>

Kampong Cham province is located about 120 kilometers northeast of Phnom Penh, the capital of Cambodia. Under the Khmer Rouge regime, people relocated to the northern part of Stoeung Trang district, Kampong Cham province were directed to clear the forest in order to establish a new settlement, the Toul Sambour Commune. Analysis of the 1984 Landsat MSS and the 1991 Landsat TM data shows substantial re-growth of forest cover at this site. With the fall of the Khmer Rouge regime in 1985, many of the people who were forced to reside in parts of the country against their will, returned to their native villages, towns and cities. This was true as well for people forced to live in Stoeung Trang district. By 1991, the abandoned area around Toul Sambour Commune had returned to forest cover.

4 The Emerging Urban Equation

The emphases on Southeast Asia’s forest cover in the science and development arenas have been driven in large part by climate change, sustainability and biodiversity questions (Cruz et al., 1998; Galloway and Mellillo, 1998; IGBP 1998; Ganeshaiah et al., 2001; Macnab and Moses, 2001; Sanchez-Azofeifa 2003). An emerging global change issue on par with forest cover is that of urban and peri-urban expansion (World Bank 1997; Kressler and Steinhocher, 1998; Glaeser and Kahn, 2003). The human dimensions effecting land cover and land use of forested areas in Southeast Asia are...
related directly or indirectly to such phenomena as rural-to-urban migration, economic development and the integration of developing countries into a global market economy sustaining industrialization and the exploitation of natural resources, and the development of infrastructure tying remote areas ever closer to urban cores and markets.

A case study of the Klang-Langat watershed, which encompasses Kuala Lumpur, Malaysia, is exemplary of the urban impacts on land cover and land use (Mastura et al., 2002). The Klang-Langat watershed is located in the mid-western part of Peninsular Malaysia and covers eight administrative districts of Selangor, Negeri Sembilan and the Federal Territory of Kuala Lumpur. The Klang-Langat watershed represents the most highly urbanised region in Malaysia. Kuala Lumpur, the capital city of about 1.5 million people, is located at the confluence of Klang and Gombak rivers. The watershed has several patches of upland forest, lowland forest and mangrove forests. Besides the built-up areas and forests, the other major land cover in the area is agricultural, of which the main crops are oil palm, natural rubber, coffee, cocoa and coconut. The Klang-Langat watershed is also vital in terms of the domestic water supply that it provides to the most densely populated area in Malaysia. There are currently four dams in this watershed: Batu Dam, Klang Gate, Pangsun Dam and Semenyih Dam.

A study of nine Landsat TM scenes of the Klang-Langat watershed measured the land cover and land use changes between 1989 and 1999 (annual data except for 1992 and 1997). In this ten-year period, the urban area grew by 159% while forest cover, agricultural and mining areas declined 18.1, 18.6 and 43% respectively. In 1989, urban land use occupied 373.8 km$^2$ but by 1996 the total coverage had expanded to 654.9 km$^2$, nearly double that from 1989. By 1999, the urban land cover in the Klang-Langat watershed encompassed 966.5 km$^2$. The new urban growth areas included the development of a new International Airport (Kuala Lumpur International Airport), the establishment of new towns (Nilai, Putrajaya, Bangi, Tun Hussin Onn), and the expansion of existing satellite urban areas. Of the ten land cover and land use classes measured over the ten-year period from 1989 to 1999, the only growth was in urban areas and water bodies (as a result of dam constructions). All other classes declined by the following amounts: mining (43%), oil palm (2%), rubber (36%), coconut (9%), horticulture (22%), dipterocarp forest (10%), mangrove (35%), bare land (8%) and grassland (21%).

Dipterocarp forest located along the upper catchment of the Klang-Langat watershed declined 10% over this time period, from 846.9 km$^2$ in 1989 to 758.5 km$^2$ in 1999. Mangrove forests areas also declined in this period from 394.6 to 258.4 km$^2$ between 1989-1999, a 35% loss. Not all of the Dipterocarp forest or mangrove forest loss can be attributed to conversion to urban land use. However, 64% of the change in Dipterocarp forest converted to either urban (30%) or rubber plantations (34%) The period saw a similar fate for mangroves: 55% of all the mangrove forests that were converted changed to urban (21%) and oil palm plantation (34%). The conversion to plantation agriculture in close proximity to a large, and economically important, market serving a global economy is not surprising and serves as an example of the types of changes the urban sphere is exacting on land cover.

The Klang-Langat watershed is an example of the spatial and temporal dynamics of a large and growing metropolitan area. We see similar land cover and land use changes in areas that are within proximity of other large metropolitan areas. The
upper Citarum watershed in West Java is a mountainous area covering approximately 2.5 x 10^6 hectares. It lies about 75 kilometers southeast of Jakarta. Within the watershed, three large-scale multipurpose dams have been constructed. The impacts of these multi-purpose dams on land cover and land use, together with the market effects of Jakarta’s proximity, have been significant. Analysis of eight Landsat TM scenes from 1989 to 1998 show a conversion of agricultural lands to urban, primary settlement and industry at an annual rate of nearly 30,000 hectares per year. The greatest change was observed between 1990 and 1991 with more than 50,000 hectares of agricultural land being converted to urban land uses (Gunawan et al., 2002).

The flood control function of the dams resulted in downstream areas becoming more suitable for settlements as the risk of yearly flooding diminished. The irrigation function of the dams also provided a steady flow of water that allowed for greater intensification of agricultural land uses. The facilities also provided hydro-electricity supporting the development of industry. While the initial efforts of the dam construction dovetailed with the expansionist agricultural policy of the early 1970s, the area witnessed land cover and land use changes in the late 1980s and 1990s tending toward urbanization as a result of the influence of Jakarta’s economic pull, the magnitude of which has increased with the development of a transportation network linking the mountainous region of the watershed to Jakarta. In this case, the sphere of urban influence on land cover and land use in the Citarum watershed is not local and spatially contiguous to the urban core as in the case of Kuala Lumpur, and the Klang-Langat watershed in Malaysia, but proximate and connected through transportation networks to Jakarta, and reinforced by infrastructural development (dam construction), market influences and national policies supporting industrialization and economic development.

5 Climate Impacts of Land Cover and Land Use Change in Southeast Asia

Much research has been aimed at increasing understanding of the human dimensions of land cover and land use change, and this continues to be an important area of discovery and analysis. There is, however, a growing effort to also understand the biophysical influences of land cover and land use change patterns over time. Certainly, some climatic factors are themselves driven by anthropogenic forces, from human impacts on the landscape (e.g. deforestation, carbon flux and climate impacts; or urban heat island effects on local climate conditions). At national, regional and global levels, there are efforts to understand and minimize the impacts of extreme climatic events, topics of grave importance in Southeast Asia’s monsoon climate. The climate extremes of heavy rains and El Niño/La Niña drought, together with periodic earthquakes and volcanic eruptions, are important influences on the land cover and land use change in the region.

A case study of the Magat watershed on Luzon Island in the Philippines documents the climatic impacts of land cover at the local scale (Crisostomo et al., 2002). The Magat watershed in Nueva Vizcaya is a forest reservation area under Philippine Proclamation 573 dated June 26, 1969. It is considered as a critical watershed and supports the Magat multi-purpose dam which is vital for irrigation, flood control, and hydroelectric power. The prevailing climate of the Magat watershed falls under two categories based on the Modified Coronas, type I and type II. In Type I climate, there are two pronounced seasons: dry from November to April and wet during
the rest of the year. In the Type II climate, seasons are not very pronounced; relatively dry from November to April and wet during the rest of the year. The southwest monsoon and the South Pacific trade winds serve as the most influential climatic controls. The rains brought about by these two air masses contribute about seventy-five percent (75%) of the annual rainfall in the area. Annually, the watershed receives about 1,400 millimeters of rainfall in low altitude areas and about 2,400 millimeters in high altitude locales. However, there is an on-going variation in climate due to the El Niño and La Niña phenomena.

The Magat River crosses the Magat watershed from south to north. The watershed is at the confluence of three mountain ranges: the Palali range located to the east, the Sierra Madre range to the south and southeast, and the Cordelleria mountains to the west and southwest. Mount Pulag with an elevation of 2,922 m. above mean sea level is the highest peak in the watershed.

Table 5: Land Cover and Land Use from 1988 - 1998 in the Magat Watershed, Luzon, Philippines

<table>
<thead>
<tr>
<th>Land Cover Type</th>
<th>Area in Hectares Covered by Each Land Cover Type Per Year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Forest</td>
<td>52,166</td>
</tr>
<tr>
<td>Secondary Forest/Tree Plantation</td>
<td>33,563</td>
</tr>
<tr>
<td>Agriculture</td>
<td>23,391</td>
</tr>
<tr>
<td>Bareland</td>
<td>730</td>
</tr>
<tr>
<td>Built-up</td>
<td>336</td>
</tr>
<tr>
<td>Grassland</td>
<td>118,143</td>
</tr>
<tr>
<td>River</td>
<td>798</td>
</tr>
<tr>
<td>TOTAL</td>
<td>231,115</td>
</tr>
</tbody>
</table>

Analysis of eight Landsat TM scenes from 1988 to 1998 (annual data except for 1989, 1991, and 1995) documents the climate effects of the La Niña phenomenon on land cover and land use in the watershed. The El Niño event of 1997 severely affected the agricultural activities in the Magat watershed. Analysis of the 1997 Landsat TM data, acquired in October (the rainy season), show the least area in non-tree agricultural use (within the 1988-1998 period of analysis), and a significant increase in open and grass land areas (see Table 5). Agricultural areas declined from 1996 to 1997 by 20,574 hectares, while grasslands increased by 30,343 hectares during this same period.

6 The Outlook for Land Cover and Land Use Change in Southeast Asia

Southeast Asia’s complex and dynamic historical, biophysical, geographical, and socio-cultural characteristics should make one pause before predicting its future land cover. That said, analyses of satellite data at the regional scale indicate a continued decline of forest cover and the expansion of agriculture and urban land uses. However, the trend in forest cover change from forest to other land cover and land use types, for the region as
a whole, appears to be leveling off. Unfortunately, land cover change is not solely about the wholesale conversion of one type or class to another. Direct observation and new methods of satellite data classification (Qi et al., 2000; Matricardi et al., 2001) are highlighting the degradation effects of selected logging on forests. In such areas, forests may remain intact as a cover type, but the quality and structure of the forest cover may be radically altered.

The outlook for land cover change in general, and forest cover specifically, at smaller scales is somewhat encouraging. There has been a growing momentum in nearly all Southeast Asian countries to curb the total exploitation of forests and to establish protected areas and national parks in order to preserve forested areas. Furthermore, countries are looking to the Kyoto Protocol to provide further incentives to reduce the rate of forest cover decline and support afforestation and reforestation activities. Our scientific understanding of upland agriculture systems and the cycles of forest conversion and abandonment are giving us new insights as to the role of land cover and land use in understanding carbon sources and sinks that impact climate change.

Key questions remain for understanding the progression of land cover and land use change in Southeast Asia, how the various proximate and distant forces of land cover and land use change operate at multiple-scales, and the eventual human and ecological impacts of these changes. Advances in GIS and remote sensing analysis as well as the development of spatially-explicit diagnostic and prognostic models will add significantly to our understanding of land cover and land use change in Southeast Asia over the next decade. With the trends in developing national spatial data infrastructures (NSDI) that support spatial decision support systems (SDSS), the acquisition and creation of new geographic data sets, and the institutional emphasis on geographic information systems for sustainable development (GIS-D), we can expect to see an exponential growth in regional contributions to our understanding of land cover and land use change, not only in Southeast Asia, but globally.

7 References


CHAPTER 8

NORTHERN EURASIA
Remote Sensing of Boreal Forest in Selected Regions

OLGA N. KRANKINA¹, GUOQING SUN², HERMAN H. SHUGART³, VYACHESLAV KHARUK⁴, ERIC KASISCHKE⁵, KATHLEEN M. BERGEN⁶, JEFFREY G. MASEK⁷, WARREN B. COHEN⁸, DOUG R. OETTER⁹, MAUREEN V. DUANE¹

¹Department of Forest Science, Oregon State University, 202 Richardson Hall Corvallis, OR 97331-5752 USA
²University of Maryland, NASA GSFC, Biospheric Sciences Branch Greenbelt, MD 20771 USA
³Department of Environmental Sciences, University of Virginia, Box 400123 Charlottesville, Virginia 22904 USA
⁴Forest Biophysics Lab, V. N. Sukachev Institute of Forest, Academgorodok, Krasnoyarsk, 660036 Russia
⁵Department of Geography, University of Maryland, 2181 LeFrak Hall College Park, MD 20742 USA
⁶School of Natural Resources and Environment, The University of Michigan, 430 E. University, Ann Arbor, MI 48109-1115 USA
⁷NASA GSFC, Code 923, Greenbelt, MD 20771 USA
⁸USDA Forest Service, PNW Research Station, 3200 SW Jefferson Way, Corvallis, OR 97331 USA
⁹Department of History and Geography, Georgia College & State University, Milledgeville, GA 31061-0490 USA

1 Introduction

Boreal forest is the major type of land cover in Northern Eurasia. It forms one of the world’s largest forest tracts and amounts to some 25% of the world’s forest cover. Boreal forest ecosystems developed under the influence of an active natural disturbance regime that created a complex and dynamic pattern of land cover. Species mix, patterns of succession, and the role of specific disturbance factors vary from region to region. Fire and insect damage represent two major natural disturbance factors, although the extent and patterns of these disturbances may partly reflect human influences. Human impact on boreal forests is expanding throughout Northern Eurasia and includes timber harvest, fire control, drainage of peat lands, urban, agricultural and infrastructure development, industrial pollution, forestation and conservation measures. The impact of projected climate change and potential feedbacks from boreal forest ecosystems are expected to be strong making it very important to understand the current patterns of forest cover, its attributes, and change over time. Case studies within the boreal forest of Eurasia focus on regions that vary greatly in the extent of human impact on forest ecosystems. In the heavily populated west (St. Petersburg region) few primary forests remain and repeated logging is the major disturbance factor while urban expansion and agricultural change (including abandonment of agricultural lands) also plays a role. In
Central Siberia, timber harvest is expanding into previously unexploited forests while fire and insects continue to play a major role as some abandonment of agricultural lands adds to forest cover. Finally, in Northeast China fire control is taking effect while intensive use of forests is maintained and localized land clearing for agriculture continues. Remote sensing was used in all three study regions as a basis for the analysis of forest cover and disturbance patterns.
2 Case Studies

2.1 ST. PETERSBURG REGION

2.1.1 Study region
The St. Petersburg region is located in the forest zone of NW Russia between 58° and 61° N and between 29° and 34° E. The region occupies 8.1 million ha of land surface and currently 53% of this area is covered with forests. The natural vegetation belongs to southern taiga type; major conifer species include Scots pine (*Pinus sylvestris* L.) and Norway spruce (*Picea abies* (L.) Karst.) both growing in pure and mixed stands. After disturbance, these species are often replaced by northern hardwoods including birch (*Betula pendula* Roth.) and aspen (*Populus tremula* L.). The climate is cool maritime with cool wet summers and long cold winters. The mean temperature of July ranges from +16° to +17° C, and the mean temperature of January is -7° to -11° C; mean monthly temperatures are negative November through March, and annual precipitation is 600-800 mm. The region is a part of the East-European Plain with elevations between 0 and 250 m a. s. l. The terrain is mostly flat and rests on ancient sea sediments covered by a layer of moraine deposits. The region was completely glaciated during the Ice Age, and contains numerous glacial features. Soils are mostly of the podzol type on deep loamy to sandy sediments. Numerous sites in the northwestern part of the region have very thin sandy soils and exposed granite bedrock, whereas the southern and eastern parts have deep silty soils and numerous expansive peat bogs. A large agricultural region stretches south and west from St. Petersburg, a city of over 5 million people (Krankina et al., 1998).

2.1.2 Mapping of Land Cover
The mapping of land cover focused on forests and selected attributes of forest cover important for modeling carbon dynamics: species composition, biomass, and age. The method for mapping forest attributes using integration of ground reference data sets and Landsat satellite imagery was first developed for the Pacific Northwest (Cohen et al., 1995; 2001) and then was adapted for the conditions of the St. Petersburg region (Oetter et al., submitted). A wealth of ground reference data was available from the Northwest State Forest Inventory Enterprise (Kukuev et al., 1997). About 75% of the forest lands in this region are under state management and are inventoried every 10 years. During the region-wide inventory of 1992-93 crews from the local inventory enterprise surveyed each forest stand polygon (a homogenous patch of forest vegetation) delineated from aerial photographs. Forest stand data for this study came from one ranger district in each of three state forests (Roschino, Lysino, and Volkhov) that were selected to represent the variation in forest cover within the region (Figure 2). Polygons were referenced to Landsat imagery and field data for each included tree species composition, age, wood volume, and characteristics of non-forest lands (bogs, clearcuts, and meadows). Biomass for each forest polygon was calculated from forest inventory data (dominant species, forest age, and wood volume) and available allometric equations (Alexeyev and Birdsey, 1998). From the total of 12,791 polygons in all three forests, about 1500 forested polygons were selected for analysis; non forest, small (<2 ha), and heterogeneous polygons were eliminated. The range of ages and biomass densities represented in field data was quite narrow: less than 1% of polygons had ages >160 years and 3% had biomass >250 Mg ha⁻¹. This is because most of the
forests in the region were repeatedly logged for timber on a rotation of about 100 years. The selection of Landsat imagery was limited initially to 1992–95 to match the time of the ground data collection. The high latitude of the region further limited the selection to the summer months (mid-May to mid-September).

Figure 2. Landsat image mosaic and ground data locations in the St. Petersburg region. The spectral reflectance data for forest pixels were used for continuous modeling of three forest cover attributes—species composition, forest stand age (years), and live forest biomass (Mg ha$^{-1}$). The training and the test set each included 735 forest polygons separated into five forest cover types, depending upon the relative dominance of forest tree species: (1) hardwood with hardwood cover ≥70%, (2) pine with pine cover ≥70%, (3) spruce with spruce cover ≥70%, (4) mixed conifers with the sum of spruce and pine cover ≥70% but neither species alone ≥70%, (5) mixed hardwood-conifer – all remaining polygons. The mean spectral signatures for polygons from these classes were extracted from a 6-band TM layer, and a supervised classification was performed on radiometrically normalized TM mosaic of the entire region to separate the forested area into these five forest classes (Oetter et al., submitted).

It proved impossible to find enough cloud-free imagery within the specified time frame to cover the entire region, so these images had to be supplemented with scenes from 1986 and 1987. Overall, 12 separate Landsat Thematic Mapper (TM) images and one Multispectral Scanner (MSS) image were acquired (Figure 2). Geometric rectification was performed by first selecting a map-registered base image (path 182, row 18 for 19 May 1992) that provided the geographic reference to which other images were geometrically corrected. Each of the 14 Landsat images was clipped to the St. Petersburg region boundary and subjected to multiple iterations of unsupervised classification, to construct a map with eight land cover classes (Agriculture, Bog,
Built/Urban, Cloud, Forest, Shadow, Shrub/grass, and Water). In addition, expert judgment of the raw imagery was used for visual reference, a political map (Kupidonova 1994) helped to identify agricultural areas, and a hand-drawn map of peatlands (Botch, unpublished) helped identify bogs. Bogs that had visual indications of human manipulation (draining and mining) were manually recoded to a separate class. The overall accuracy of this map was assessed to be 88% (Oetter et al., submitted). The total area of forest derived from our land cover map (5.57 million ha) is higher than forest area reported by forest inventory system (4.92 million ha). The difference is attributable to exclusion of several types of tree-covered lands from forest inventory (e.g. parks, resorts, small woodlots around summer homes). The total area of mapped bogs (0.73 million ha) is in good agreement with the total bog area reported by forest inventory (0.69 million ha; Filimonov et al., 1995).

Table 1. Multispectral indices for predictive modeling of live forest biomass (Standardized TM Band Loadings) and model test results.

<table>
<thead>
<tr>
<th>Forest type</th>
<th>Coefficients$^2$</th>
<th>R$^3$</th>
<th>RMSE Mg ha$^{-1}$</th>
<th>Number of test polygons</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>TM2</td>
<td>TM3</td>
<td>TM4</td>
<td>TM7</td>
</tr>
<tr>
<td>Hardwood</td>
<td>-0.26</td>
<td>0.40</td>
<td>0.34</td>
<td>0.77</td>
</tr>
<tr>
<td>Hardwood/conifer mix</td>
<td>-0.26</td>
<td>0.62</td>
<td>0.58</td>
<td>0.46</td>
</tr>
<tr>
<td>Mixed conifers$^1$</td>
<td>-1.43</td>
<td>0.83</td>
<td>-0.09</td>
<td>-0.18</td>
</tr>
<tr>
<td>Pine$^1$</td>
<td>-0.10</td>
<td>0.11</td>
<td>0.39</td>
<td>0.80</td>
</tr>
<tr>
<td>Spruce$^1$</td>
<td>0.74</td>
<td>-0.80</td>
<td>-0.39</td>
<td>1.22</td>
</tr>
</tbody>
</table>

1 For Lysino and Volkhov sites only
2 Canonical correlation coefficients for squared value of the TM band reflectance.
3 Correlation between multispectral index and observed forest biomass in test data set.

Within each forest cover type, predictive models relating spectral values to forest age and live biomass were generated using a ‘reduced major axis’ (RMA) regression approach (Curran and Hay, 1986). This method solves a regression equation by minimizing error in both spectral data and the forest attribute, so that predictive bias is reduced (Cohen et al., 2003). RMA requires a single independent variable, so the six TM bands were collapsed into one canonical correlation analysis (CCA) index that was calculated for a selected subset of the TM bands (Table 1). The TM data values were transformed to meet the CCA assumption of a linear relationship between the vegetation attribute and each explanatory variable. The squared value of the TM band reflectance produced the best results. The correlation of resulting multispectral indices with forest age and biomass was examined on the test set of polygons. It was statistically significant (at $\alpha = 0.05$) in all cases with higher correlation coefficients for biomass (Table 1) than for age ($R = 0.19$ to $0.59$). Other studies of forest attribute prediction with Landsat TM data found similar accuracy (Hyyppaa et al., 2000). RMSE for individual forest types ranged between 35 and 51 Mg ha$^{-1}$ and between 15 and 30 years for biomass and age, respectively. Even though forest age and biomass correlate with each other, the canonical correlation coefficients for these two attributes were generally different. The regional total of forest biomass modeled as described above agrees well with the estimate based on forest inventory data (542 Tg and 538 Tg,
respectively) and so did the regional average age for conifers (71 and 68 years) and for hardwoods (44 and 50 years) (Oetter et al., submitted; Treyfeld et al., submitted).

Results show that forest attributes including species composition, biomass, and age can be successfully modeled using Landsat imagery but models require extensive ground data and appear to be strictly local. Multispectral indices were developed for each forest type individually and conifer-dominated types had to be further subdivided geographically to isolate the northwestern part of the region with shallow rocky soils. Model extrapolation beyond the boundaries of the St. Petersburg region is problematic, but the approach can be replicated in other regions of Northern Eurasia because similar ground data exist for about half of all forests lands in Russia (Kukuev et al., 1997) and in China as well.

2.2 CENTRAL SIBERIA

2.2.1 Study Region
The vast land area of Siberia is subdivided geographically into West Siberia, East Siberia, and the Far East. The study region reported on here is primarily in East Siberia, but also includes portions of West Siberia (west of the Yenisei River), and for this reason we call the region Central Siberia. Central Siberia ecoregions include arctic deserts, tundra, and forest-tundra in the north; taiga forests in the central part; and steppe, forest-steppe, and productive agricultural areas in the south. Our study of Land-Cover Land-Use Change (LCLUC) to date focused on the dynamics of the taiga forests in three territories: Tomsk Oblast, Krasnoyarsk Krai, and Irkutsk Oblast (Figure 1; 50°N - 65°N and 85°E - 115°E). Within each of these territories we have established Landsat-sized (185 x 185 km) case study sites in which we have mapped and studied forested ecosystem dynamics occurring over the 25-year timeframe, 1975-2000.

The climate in the sites is continental with short hot summers and severe winters. The January mean temperature is ~–22°C, and the July mean temperature is ~+18 °C (Popov 1982). Fifty to sixty percent of precipitation falls between July and September, and snow cover begins in October and remains for seven months. Physiography includes the Central Siberian plateau dissected by major rivers. Mountainous areas are present in the Irkutsk case study site. Soils consist of sands and clays. During wintertime, soils may be frozen to a depth of 2 - 3 meters, and there are local cases of permafrost.

The forest in the case study sites is a mosaic of several communities at various age and disturbance stages. The ‘light-needle’ coniferous forest includes Scots pine (*Pinus silvestris*) and larch (*Larix sibirica*). These light-dependent species grow in a range of terrain from xeric to mesic. The ‘dark needle’ forests of spruce (*Picea obovata*), fir (*Abies sibirica*), and Siberian pine (*Pinus sibirica*) grow on richer or moister substrates, on slopes of northern exposure, or as accompanying species in “light-needle” dominated communities (Bondarev 1990). Deciduous birch-aspen (*Betula pendula, Populus tremula*) forest and mixed pine-deciduous-larch forests are present on post-disturbance sites.

Human settlement in the region started in the 17th century. Timber clear cuts, periodic insect outbreak, and fire dominate the contemporary disturbance landscape. Logging started intensively in the 1940s and ‘50s staffed by prisoners of the GULAG operations, and the most intensive logging of this type occurred in the 1960s and ‘70s.
The main harvested species is pine (*Pinus sylvestris*), and second is larch. Beginning in the early 1990’s, large areas of forests in Eastern Siberia have been affected by outbreaks of the Siberian silkworm (*Dendrolimus sibericus*) (Environmental Working Group). Insect infestation can lead to the death of forest stands and increased susceptibility to fire. Fires, fuelled by the debris left from the clear-cutting of trees and insect-related tree mortality, are enhanced by climate warming and human activities, and may be increasing in frequency and intensity (Kasischke et al., 1999). Pollution from mineral and paper processing and power plants is also an increasing concern.

### 2.2.2 Mapping of Land Cover

Our experience with multi-spectral 30 meter Landsat data (and Russian data at similar scales) has demonstrated that it is particularly well suited for documenting spatial and temporal patterns resulting from logging, fire, insect damage, and agricultural abandonment. A combination of Landsat visible, infrared, and thermal bands provides adequate spectral resolution for detection of desired classes. Level I land-cover classes in the region are: forest, agriculture, urban, wetland, bare, and water. Level II forest classes include up to four mature forest classes (conifer, mixed, deciduous, and larch); and four forest disturbance classes (recent fire, recent cut, insect infestation, and young regeneration). The goal of the LCLUC analysis was to look at change over the time period 1975-2000 using time-series of Landsat imagery. The best time-series datasets for case study sites included a MSS image from the mid-1970s, a TM image from approximately 1990 and an ETM+ image from approximately 2000.

#### Table 2. Land-cover percentages in 2000 for each of the three case study sites.

<table>
<thead>
<tr>
<th>Forested Land Cover</th>
<th>Tomsk 147-20</th>
<th>Krasnoyarsk 141-20</th>
<th>Irkutsk 133-23</th>
</tr>
</thead>
<tbody>
<tr>
<td>Conifer</td>
<td>15.51</td>
<td>8.57</td>
<td>22.28</td>
</tr>
<tr>
<td>Deciduous</td>
<td>11.77</td>
<td>23.82</td>
<td>6.48</td>
</tr>
<tr>
<td>Mixed</td>
<td>28.16</td>
<td>34.96</td>
<td>12.13</td>
</tr>
<tr>
<td>Larch*</td>
<td>NA</td>
<td>NA</td>
<td>15.83</td>
</tr>
<tr>
<td>Recently Cleared</td>
<td>8.30</td>
<td>4.38</td>
<td>3.41</td>
</tr>
<tr>
<td>Recently Burned</td>
<td>0.00</td>
<td>1.77</td>
<td>5.88</td>
</tr>
<tr>
<td>Insect Damaged</td>
<td>0.00</td>
<td>7.76</td>
<td>0.00</td>
</tr>
<tr>
<td>New Regrowth</td>
<td>21.09</td>
<td>8.62</td>
<td>2.26</td>
</tr>
<tr>
<td><strong>Total Forest</strong></td>
<td><strong>84.83</strong></td>
<td><strong>89.88</strong></td>
<td><strong>68.27</strong></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Other Land Cover</th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Agriculture</td>
<td>3.64</td>
<td>7.95</td>
<td>12.92</td>
</tr>
<tr>
<td>Wetland</td>
<td>10.72</td>
<td>1.85</td>
<td>15.11</td>
</tr>
<tr>
<td>Urban</td>
<td>0.80</td>
<td>0.32</td>
<td>0.59</td>
</tr>
<tr>
<td>Bare</td>
<td>NA</td>
<td>NA</td>
<td>3.11</td>
</tr>
<tr>
<td><strong>Total</strong></td>
<td><strong>100.00</strong></td>
<td><strong>100.00</strong></td>
<td><strong>100.00</strong></td>
</tr>
</tbody>
</table>

* larch does not form pure stands in these (NA) areas

Landsat pre-processing included georectification (using Landsat-7 ETM+ definitive ephemeris and georectifying TM and MSS to the ETM+), haze reduction, and
Land-cover classifications were created at the highest resolution possible for each time period, using the land-cover classes listed above. Supervised classification and classification validation was possible using ground-truth data from Russian researchers at the Sukachev Institute of Forest (Krasnoyarsk) transmitted via interpreted images and ancillary data. Table 2 lists the present (2000-2001) percentage of land-cover in three case study sites derived from our ETM+ classifications. Post-classification land-cover change was calculated at two time intervals. The use of three dates is a key factor in successful change detection. Some categories, particularly forest disturbances, are difficult to label when a single date classification is done, and therefore, a three-date change trajectory approach was developed (Miller et al., 2002).

2.2.3 Patterns of Forest Disturbance and Regrowth

Analyses of statistical data from Russian agencies at national and territorial levels document a steady high rate of logging from 1965 to 1990 and subsequent sharp decrease starting in 1990, coinciding with the dissolution of the Soviet Union and its state-run and subsidized economy. Logging activity has started to increase in the past several years with new private enterprises, but to date only very slightly (Figure 3).

![Figure 3. Trends in total wood removal ($10^6\text{m}^3$) for the Russian Federation and for each of the three territories of the Central Siberia case study sites. Sources: Pre-1991 data are from Forest Complex of USSR, 1991, All Union Scientific Research Institute of Economics. Post-1991 data are from Industry of Russia, Goskomstat of Russia.](image)

Statistics derived from remote sensing show a similar pattern and also shed light on additional characteristics. A high resolution Corona image from the Krasnoyarsk case study site from 1965 showed that a high percentage of the forested landscape had been logged or burned by that date. This is also evident from the amount of forest in early secondary succession in the 1974 Landsat MSS image. Over the 25-year Landsat analysis of the Krasnoyarsk case study site, the recently logged class declined from 10% to 4% in the first 16 years (1974-1990) and subsequently remained constant. Due to decreased logging, the area of young forest regrowth in the case study site increased from 17% to 24% during the first 16 years (1974-1990) and decreased to 9% afterward (1990-2000). Total forest cover increased by only 1% (from 57% to 58%) between
These remote sensing analyses also show that specific forest communities are changing. In the Krasnoyarsk case study site, coniferous forest has been reduced continuously over the period 1975-2000, while deciduous and mixed forests have increased. Preliminary investigations show that a similar trend exists in the Tomsk and Irkutsk case study sites.

In the Krasnoyarsk study site, fire may have accounted for less than 2% of total area of change over the period 1974-2000. However this is not entirely conclusive as some fires that occurred between the image dates may have been missed. Severe insect infestation broke out in 2000 in the Krasnoyarsk site affecting 8% of total land cover. Although insect damage was not found in our LCLUC Tomsk and Irkutsk sites, it is present in the larger territories.

A somewhat unexpected finding was the conversion of agricultural lands to forest regrowth. Beginning even before the dissolution of the government-run economy, many areas assigned to collective farms began to be vacated. Once abandoned, these agricultural areas acquire shrub cover, eventually succeeding to young deciduous and pine trees. This is beginning to become apparent on the Landsat imagery, although distinguishing recently abandoned agricultural fields from annual phenological change or crop cycles is somewhat difficult using spectral remote sensing data. Careful assessment of the Krasnoyarsk data shows that agricultural use of the landscape may have been reduced from 14% to 8% gradually over the study period 1974-2000 largely through abandonment of collective farming areas (Bergen et al., 2003). One hypothesis from our LCLUC study is that the change-over from the government-run economy to a market-based system among the nations of the old Soviet Union will leave different “footprints” on the landscape, and this first – and yet early – analysis suggests that this is indeed the case.

2.3 NORTHEAST CHINA

2.3.1 Study Region
Northeast China includes three provinces: Liaoning, Jilin and Heilongjiang, and the northeastern part of Inner Mongolia Autonomous Region (Figure 1D). Together, these areas account for about 30% of the forested land in China, and include all of the Boreal forest in China. Geographically, the forested lands occupy a mountainous perimeter that rings the agricultural Manchurian (Northeast) plain. The west perimeter includes the Da Xingan (Greater Xing’an) Ling Mountains, which extend from north to south through western Heilongjiang and the Inner Mongolian region. The climate is cold, and the dominant species is larch (Larix gmelini), with lesser populations of white birch (Betula platyphylla), Mongolia oak (Quercus mongolica), and David poplar (Populus davidiana). To the east, the Xiao Xingan (Lesser Xing’an) Ling and Chang Bai Mountains extend through central Heilongjiang and eastern Jilin provinces. In this warmer, wetter region mixed evergreen-needleleaf (Korean Pine, Firs, Japanese Yew) and broadleaf (Amur Linden, Maple birch, White birch, oak) forests are common. Larch (Larix olgensis) have been planted throughout the division as a major forest crop. The southernmost perimeter of the region includes the Qian Mountains in Liaoning province, where the main forests are mixed broadleaf deciduous, with abundant oaks, maples, and birch.
The forests in Northeast China have been undergoing dramatic changes during the last several decades due to human activities and forest management. As in other areas of Eurasia, wild fires have historically played a major role in Northeastern China. For example, a forest fire, which burned from 6 May to 4 June 1987, destroyed more than one million ha of forest and nearly 25 million cubic meters of timber in this area (Goldammer and Di, 1990). Following the 1987 fire, the government and local forest bureaus made fire prevention and suppression a priority. During the 13 years after the fire the Da Xingan Ling Forest Bureau invested 350 million yuan to build 127 observation towers and an AVHRR receiving station, and financed mobile fire patrol teams. The average area affected by forest fire annually was reduced from 1.9% before 1987 to 0.027% during 1988-2000. Thus, while fire is a natural disturbance process in the Boreal forest, current management has dramatically suppressed the frequency of fire in Northeastern China.

Similar changes have occurred with respect to forest clearing. As in other parts of China, significant declines in forest cover occurred throughout the mid-20th century, as a result of agricultural conversion and logging. A major cause of deforestation during the last century has been commercial lumber production, although other human activities, such as slash and burn farming, unauthorized felling of trees, the culture of silkworms and edible fungi, and construction of railroads, highways, and high voltage power lines have also contributed to forest destruction. Following the disastrous floods of 1998, the Chinese government established the Natural Forest Conservation Program (NFCP) to mitigate future land degradation. Marginal farmland will be mandated to convert to forest land with government compensation. In addition, clearing of forests in many sensitive watersheds has been banned. This policy may be a turning point for forest sustainability in China (Zhao and Shao, 2002). In addition, since 1978 the Three Norths shelterbelt project has sought to protect agricultural land throughout northern China by planting trees to act as windbreaks and reduce erosion (Yu 1990).

Although the scope of changes to Chinese forests during the last century has been large, but their effect on carbon fluxes has been only roughly quantified (Fang et al., 2001). Our study of forest cover in Northeast China from remote sensing data was carried out as part of the CEOS Global Observation for Forest and Land-Cover Dynamics (GOFC-GOLD) program and included mapping of forest cover with Terra-MODIS data, forest-cover dynamics from Landsat-5 and Landsat-7 data.

2.3.2 Forest Cover Mapping with TERRA-MODIS Data

Although several papers have used multi-temporal AVHRR (Liu et al., 1998; Young and Wang, 2001) or SPOT-4 Vegetation (Xiao et al., 2002) data, forest maps with a pixel size less than 1km have not yet been produced for entire region. In this study, we processed MODIS reflectance data to generate forest cover and type maps at a spatial resolution of 500 meters.

MODIS 500-meter resolution, 16-day NDVI composites from June 2000 to January 2001 (MOD13A1 product) were used to classify forest extent (Figure 4). The MODIS product from 2000 and 2001 often contained bright and/or dark artifacts. To rectify these anomalies, a ‘smoothing’ method (Xiao et al., 2002) was used. The temporal NDVI data were classified into 255 clusters using ISODATA with 20 iterations. These clusters were then labeled by referring to 1:50,000 forest maps, Landsat-7 images, the Atlas of Forestry in China (Forest Ministry of China 1990), and the Vegetation Atlas of China (Editorial Board of Vegetation Map of China 2001).
Figure 4. Land-cover classification derived from MODIS NDVI time series. (see CD for color image)

Land use maps from Landsat-7 ETM+, prepared by the Northeast Institute of Geography and Agricultural Ecology, Changchun China (Zhang 2002), were used to evaluate the accuracy of the MODIS forest cover classification. The classification accuracies were evaluated in terms of the forest cover areas, and found that the classification accuracies are 92.63%, 79.01%, 69.00%, and 87.87% for evergreen needle, deciduous needle (larch), deciduous broadleaf, and needle-broadleaf mix, respectively. The overall accuracy for forests is 79.76%. We also used the average mutual information (AMI) (Finn 1993) to compare the forest distribution patterns between the two maps. AMI is expressed as a percentage of the entropy of one map, representing how much information in the map can be predicted by another map. The results show that the MODIS classification and the L-7 ETM+ land-use maps share 86.4% of information on forest distribution patterns.

Table 3 summarizes the percent forest cover of provinces from different sources. The Ministry of Forestry data were derived from field surveying and aided by airphoto interpretations. In the 1999/2000 land use map from Landsat images, in addition to the forest listed in the table, there were other categories such as ‘other forests’ and dense shrubs, which amount to 7-12% of the land in Northeastern China. It was found that the discrimination between young forests and shrubs in MODIS data was difficult. This could explain discrepancies in observed forest cover between low- and high-resolution data sources.
Table 3. Percent forest cover by Provinces. The total area used for Heilongjiang, Jilin, and Liaoning Provinces were 454,440, 188,845, and 145,764 km$^2$ (Forest Ministry of China 1990).

<table>
<thead>
<tr>
<th>Province</th>
<th>Ministry of Forestry (1990)</th>
<th>1999/2000 Land Use Map</th>
<th>SPOT VEG Map</th>
<th>MODIS 500m Map</th>
</tr>
</thead>
<tbody>
<tr>
<td>Heilongjiang</td>
<td>34.4</td>
<td>38.7</td>
<td>46.9</td>
<td>48.8</td>
</tr>
<tr>
<td>Jilin</td>
<td>33.0</td>
<td>37.7</td>
<td>48.4</td>
<td>43.1</td>
</tr>
<tr>
<td>Liaoning</td>
<td>27.0</td>
<td>26.6</td>
<td>31.4</td>
<td>25.2</td>
</tr>
</tbody>
</table>

1 Forest Ministry of China 1990; 2 From Table 1, Xiao et al., 2002

2.3.3 Forest Dynamics from Landsat

Two approaches were used to examine rates of forest cover change during the 1990’s. First, the land-use maps for 1990 and 2000 were developed by hand-digitizing Landsat-5 and Landsat-7 multispectral images. The original 30-meter resolution maps were aggregated to a resolution of 25 Ha (500 x 500 m), with the numeric value of each pixel recording the percentage of forest-cover within each 25 Ha cell (Figure 5a).

Subtracting the two images reveals that forest area in Northeastern China decreased by about 1.8% during the 1990’s (Figure 5b). Forest-cover losses were most pronounced at the interface between the Manchurian plain and its mountainous perimeter. Expansion of agriculture from the central plain to the foothills, fueled partly by population migration from Southern China, has led to significant clearing in these regions. Large-scale clearing is particularly pronounced along the eastern edge of the Da Xingan Ling mountains, where entire hillsides have been deforested to make way for farming. In contrast, forest expansion is concentrated in the Chang Bai and Longgang mountain regions near the North Korean border. This region, which includes the Chang Bai Forest Preserve, is a locus of plantation forestry, where small plots of hardwoods and softwoods are continuously harvested and reseeded. Thus, there are
systematic differences across the study area both in terms of the total area cleared, as well as the mean patch size of clearing.

As an alternative to the land-use map analysis, we have independently carried out radiometric change detection for a subset of the Landsat imagery in the Northeast China study region. Orthorectified Landsat-5 TM and Landsat-7 ETM+ images, roughly spanning the 1990-2000 epoch, were calibrated, processed to surface reflectance, and then radiometrically rectified using histogram matching (Schott et al., 1988). Radiometric change detection was carried out for each pixel, with “change” being discriminated by using the spectral angle between the two dates for that particular pixel (Sohn and Rubello, 2002). Deforestation and regrowth/afforestation were separated according to the direction of the spectral trajectories in the visible and shortwave-infrared bands. As an initial step, this analysis has been carried out for two regions: the Chang Bai mountains near the Korean border, and the Da Xingan Ling range near the border with Russia and Mongolia. Initial validation of the radiometric change algorithm based on visual inspection of small training regions indicate producer’s and consumer’s accuracies of 79% and 62%, respectively for deforestation, and 93% and 97%, respectively, for the “no change” class.

The net change calculated from both methods agrees well for both the Da Xingan Ling and Chang Bai regions (Table 4). Again, the Chang Bai region shows significantly lower rates of forest loss than the Da Xingan Ling region. However, the specific areas of forest loss and gain differ between the methods for the Chang Bai region, with the land-use map analysis suggesting greater rates of both clearing and regrowth than found from our radiometric analysis. Based on direct inspection of the imagery, we feel that the radiometric analysis provides a more accurate view of forest cover change, and that the land-use maps may suggest greater activity (both clearing and regrowth) than actually took place during the 1990’s.

Table 4. Comparison of spectral-angle (radiometric) change detection with the land-use map analysis for the Da Xingan Ling and Chang Bai regions. All values are area of change (gain, loss, and net) in thousands of hectares.

<table>
<thead>
<tr>
<th></th>
<th>Spectral Angle Analysis</th>
<th>Landuse Map</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Gain</td>
<td>Loss</td>
</tr>
<tr>
<td>Da Xingan Ling</td>
<td>34.7</td>
<td>-133.8</td>
</tr>
<tr>
<td>Chang Bai</td>
<td>33.8</td>
<td>-59.7</td>
</tr>
</tbody>
</table>

In summary, it appears that during the decade of the 1990’s forest cover in Northeastern China declined slightly by 0.2% per year. Significant deforestation has occurred locally, however, as a result of agricultural conversion within foothills along the perimeter of the Manchurian plain. It should be noted, however, that the results presented here only partly reflect the recent National Forest Conservation Program, which has led to decreased harvests in the Northeastern region since 1998. A follow-up study is required to assess the effects of the NFCP on forest cover since then.
3 Discussion

Historically fire was the dominant disturbance factor over the entire territory of Northern Eurasia. Presently logging has replaced fire as the dominant disturbance factor to a different degree in all three case studies. In different parts of the St. Petersburg region the area of recent clearcuts ranges from 0.3% to 1.3% of the total forest area while burned area is only about 0.1% (Krankina et al. 2001). In Central Siberia in 2000, the recently logged area is between 5 and 10% of forest area in examined study sites. Fire was the greatest disturbance factor in the Irkutsk case study site (Table 2), while in the other two case study sites (Tomsk and Krasnoyarsk) the most current disturbance came from logging activity. In Northeastern China, fire suppression since the 1980’s proved to be particularly effective and reduced the impact of fire to extremely low levels similar to those in the St. Petersburg region. Logging rates in Northeastern China, however, were similar to those in Siberia (7.5% of forest cover loss between 1988 and 2000, most of it due to clearcut harvest). In the future, fire suppression combined with recently imposed restrictions on logging will alter significantly disturbance regimes in Northeastern China with unknown effects on ecosystem structure and composition.

The varied proportions of disturbed forest areas in the study regions reflect the net effect of forest disturbance and regeneration processes. In the Krasnoyarsk study site some of the burned areas fail to regenerate, due probably to repeated fires, insect outbreaks, or poor soil conditions. In the St. Petersburg region and in Northeastern China alike, long-term failure of forest regeneration would be a rare exception because of active reforestation programs in both regions.

Socio-economic driving forces created divergent patterns of timber harvest in the 1990’s in Russia and China. During Russia’s transition to a market economy in the early 1990’s, logging was reduced to one-half of the previously logged area (Figure 3). Decreasing logging activity contributed to shrinking the area of forest re-growth. In addition, economic decline contributed to further abandonment of agricultural lands that started even prior to the transition period. Most of the abandoned farmlands were developing into young deciduous forests. Conversely in China, the case study suggests that agricultural conversion remained extensive during the 1990’s, driven partly by immigration from the poorer regions of southern China. However, the logging reductions associated with the Chinese National Forest Conservation program will likely reverse the observed decline in forest cover in the coming years.

Remotely sensed data in combination with ground observations and existing maps proved effective in the analysis of forest cover, its many important attributes, and its change over time in all three study regions. The three case studies highlight different aspects and varied applications of LCLUC research: the St. Petersburg case study focused on mapping the attributes of forest cover that are needed for spatial modeling of carbon dynamics; the Central Siberia case study emphasized mapping of land-cover change as a basis for modeling the effects of socio-economic change on land cover dynamics; in the Northeast China case study the primary goal was to map current forest cover and track the changes during the 1990's as a basis for a GIS-based monitoring system designed to update forest cover maps in the future.
Acknowledgements. We acknowledge the following contributors to research presented in this chapter: in the St. Petersburg case study – Rudolf Treyfeld and Ewgenii Povarov of Northwestern State Forest Inventory Enterprise, St. Petersburg, Russia, and Thomas Maiersperger and Gody Spycher of Oregon State University; in the Central Siberia case study – Lara Peterson, Tingting Zhao, Bryan Emmett, Daniel Brown, The University of Michigan, Ann Arbor, MI USA; Elena Fedotova, V.N. Sukachev Institute of Forest, Krasnoyarsk, Russia; Yuri Blam and Olga Mashkina, RAS Institute for Economics, Novosibirsk, Russia; Leonid Vaschuk, Ministry of Natural Resources, Irkutsk, Russia; John Colwell, Veridian, Ann Arbor, MI USA; Nicole Miller, Satellite Imaging Corporation, Houston, TX USA.

4 References


1 Introduction

Canada and Alaska occupy an area of 11.1 million km$^2$, almost 10% of the vegetated cover of the Earth's surface. In the Western Hemisphere North of 50$^\circ$ N, terrestrial interactions with the climate system are dominated by the land mass of Canada and Alaska. The forests of this region, which occupy an area of approximately 4 million km$^2$ (~10% of global forest area), represent a wood resource of global economic significance with Canada responsible for approximately 11% of global industrial roundwood production in the 1990s (Perez-Garcia, 2002). Land cover in Canada and Alaska has been undergoing substantial changes in recent decades (Kurz and Apps, 1999; Stocks et al., 2000; Sturm et al., 2001; Silapaswan et al., 2001; Podur et al., 2002; Lloyd et al., 2003a). Between 1920 and 1970 the average stand age of forests in Canada increased from less than 60 years old to 81.5 years old (Kurz and Apps, 1999). From the forest age class distribution in Canada in 1920 (Kurz and Apps, 1999) and in Alaska in the 1960s (Yarie and Billings, 2002), it can be inferred that there were large-scale synchronous disturbances that occurred in the 1860s throughout Canada and Alaska. These disturbances have substantially influenced the structure of forests in the latter half of the 20$^{th}$ Century. Important changes in land cover that have recently occurred and that may continue to occur in the region include abrupt changes associated with disturbance (e.g., fire, insect outbreaks, timber harvest, and agriculture), and more subtle changes in vegetation dynamics that influence seasonal functioning (e.g.,
changes in the timing of soil freezing and thawing) or the spatial distribution of vegetation (e.g., the advance of tree line in regions now occupied by tundra and changes in the area of surface waters). Because of the large area occupied by Canada and Alaska, it is important to understand how and why land cover is changing in the region, and the implications of these changes for the climate system and for the ecological, economic, and cultural sustainability of the region.

There are three major pathways through which land cover changes in Canada and Alaska may influence the climate system: (1) water/energy exchange with the atmosphere, (2) the exchange of radiatively active gases (e.g., carbon dioxide and methane) with the atmosphere, and (3) delivery of fresh water to the Arctic Ocean. The exchange of water and energy between the terrestrial surface and the atmosphere has implications for regional climate that may influence global climate, while the exchange of radiatively active gases and the delivery of fresh water to the Arctic Ocean are processes that could directly influence climate at the global scale (McGuire et al., 2003a). Land cover changes also have implications for the ecological, economic, and cultural sustainability of Canada and Alaska. The northward movement of the treeline (Lloyd et al., 2003a) and the increase in tundra shrubs (Silapaswan et al., 2001; Sturm et al., 2001) associated with warming climate in much of the region has the potential to eliminate some tundra types from the Canadian Archipelago in northern Canada if climate warming continues (Gould et al., 2003). Land cover changes have implications for human livelihoods in Canada and Alaska and elsewhere through effects on subsistence resources, commercial timber resources, infrastructure, and industrial activity (e.g., oil and gas development).

In this chapter we present our current understanding of the dynamics and controls for important avenues of land cover change in the region including fire, insect disturbance, human land use, vegetation dynamics, and changes in the area of surface waters. We extend our discussion of controls to evaluate how land cover change might respond to possible future changes in the region. We then discuss the implications of future changes for major feedbacks to the climate system. Finally, we discuss the future challenges to understanding the patterns, controls, and consequences of land cover change in Canada and Alaska.

2 Dynamics, Controls, and Future Responses of Land Cover Change in Canada and Alaska

2.1 FIRE DISTURBANCE

In Canada and Alaska, fire represents an important disturbance that is much more prevalent in boreal forests than in tundra and temperate forests of the region (McGuire et al., 2002; Stocks et al., 2002; Kasischke et al., 2003). Fires in the boreal forest are responsible for about 94% of the annual area burned in Canada. While only about 3% of the fires in Canada are larger than 200 ha, these fires account for 97% of the area burned (Stocks et al., 2002); a similar pattern exists in Alaska. On average over the last 50 years, approximately 2 million ha burns annually in Canada (Amiro et al., 2001) and about 250,000 ha burns annually in Alaska (Kasischke et al., 2003), a difference that is largely explained by the larger area of boreal forest in Canada relative to Alaska. Interannual variability in the amount of area burned ranges between approximately 0.5 and 7
million ha in Canada (Amiro et al., 2001) and between <0.1 and 1.7 million ha in Alaska (Kasischke et al., 2003).

The paleorecord of fire for boreal forests in North America provides substantial evidence for linkages among fire, climate, and vegetation (Clark, 1988; Carcailliet et al., 2001; Flannigan et al., 2001). In comparison with the current fire regime, there is evidence that fire frequency was higher in Canada approximately 6000 years ago during the mid-Holocene when conditions were warmer because of greater insolation associated with variation in the earth’s orbit (Kutzbach et al., 1993). However, evidence from Alaska suggests that there are important interactions between climate and vegetation that may affect the fire regime. Although Alaska was warmer in the mid-Holocene when forests were dominated by white spruce, fire frequency was relatively low (Lloyd et al., 2003b). Fire frequency increased in Alaska about 5000 years ago during a time when the climate was becoming cooler and wetter, which provided conditions for the migration of black spruce in interior Alaska (Lynch et al., 2003). From a millennial temporal perspective, it appears that presence or absence of black spruce, which is more flammable than white spruce, may play a larger role than climate in affecting the fire regime. In contemporary Alaska, the fire cycle decreases as the percent of tree cover increases among ecoregions in the state (Kasischke et al., 2003).

Analyses of the fire regimes in Canada and Alaska during the latter half of the 20th Century have established that inter-annual variability in area burned is linked to inter-annual variability in climate for Alaska (Hess et al., 2001) and for provinces of Canada (Flannigan and Harrington, 1988). In Alaska, the area burned in a particular year is positively correlated with mean June temperature and negatively correlated with the depth of snowpack near the end of winter (Duffy et al., 2003; McGuire, unpublished). Approximately 68% of the total area burned in Alaska from 1940 to 1998 occurred during 15 years characterized by a moderate to strong El Nino, conditions in interior Alaska with above normal temperatures throughout the year and normal precipitation from February to August (Hess et al., 2001). Among different ecoregions of Alaska, fire cycle decreases with increasing growing season temperature, decreasing growing season precipitation, and increasing lightning frequency (Kasischke et al., 2003). In Canada, area burned is related to blocking highs in the westerlies that cause long sequences of days with low rainfall and relative humidity below 60%, and the occurrence of fire within a month depends on rainfall frequency, temperature, and relative humidity (Flannigan and Harrington, 1988). The proximate linkages of these conditions to fire are to lightning as an ignition source and to moisture content of forest floor organic matter, which is important for the lateral spread of fire.

In the last half of the 20th Century, the fire frequency in northwest Canada increased substantially during the 1970s and 1980s in association with a warming climate (Kurz and Apps, 1999; Stocks et al., 2000; Podur et al., 2002). While the fire frequency in Alaska has not been demonstrated to have statistically increased during the late 20th Century, there have been four large fire years since 1980 in comparison with two large fire years between 1950 and 1980 (Kasischke et al., 2003). Together, these analyses indicate that fire frequency in the Canada and Alaska region has increased in the late 20th Century and has the potential to increase in response to continued climate warming. Analyses of the response of fire weather to projections of future climate change suggest that climate change has the potential to substantially increase fire frequency throughout much of the boreal forest in North America (Wotton and
Humans affect fire regimes in the boreal forest through the ignition of fires, the suppression of fires, and land use that alters the spatial distribution of fuels. In Canada, human-caused ignitions are more prevalent in the more heavily populated southern area of the boreal ecozone. In Alaska, lightning ignitions are largely distributed throughout the interior between the Brooks and Alaska Ranges, but human-caused ignitions are largely confined to the road network (Gabriel and Tande, 1983; Kasischke et al., 2003). Although humans are an important source of fire ignition in boreal forests, they are not responsible for the majority of area burned. In Alaska, humans cause more than 60% of all ignitions, but these ignitions result in only 10% of the area burned (Kasischke et al., 2003). A similar situation exists in Canada as 80% of the area burned is attributed to fires ignited by lightning. The low amount of area burned from human-caused fires is the result of poor burning conditions at the time of ignition and responsive fire suppression efforts, as the fires started by humans are generally quickly detected and accessed because they are located in proximity to transportation networks. In Alaska, recent analyses indicate that fire cycles are longer near population centers (Chapin et al., 2003).

While fire suppression is important in promoting longer fire cycles near population centers, the alteration of fuels by human land use likely plays a role as well. For example, near Fairbanks there was substantial harvesting of spruce forests during the settlement period in the early 1900s. Forest succession has led to the replacement of spruce forests in many areas by aspen and birch forests, which are less flammable than spruce forests and inhibit the spread of fire. The combination of fire suppression and spatial alteration of fuels is presumably responsible for a fire cycle near Fairbanks that is estimated to be four times longer than the fire cycle of areas in interior Alaska located away from population centers. Finally, the fire regime may be influenced by insect infestations as stands affected by insect disturbance may be more vulnerable to fire because tree mortality generally increases the flammability of forest stands.

2.2 INSECT DISTURBANCE

On average, insect infestations annually affect a similar amount of area as does fire in the forests in Canada (Kurz and Apps, 1999) and Alaska (Werner et al., 2003). Since approximately 1920, between 1 - 2 million ha of forests in Canada have annually experienced insect-induced stand mortality (Kurz and Apps, 1999). Insects responsible for major infestations in Canada include the eastern spruce budworm, the hemlock looper, the Jack pine budworm, and the Mountain pine beetle. Records in Alaska indicate that insect infestation affects approximately 300,000 ha annually (Werner et al., 2003), but the effective area of insect-induced stand mortality is less than 300,000 ha because successive years of attack are required to cause stand mortality and only a portion of the stands affected are host species for the attacking insect; there has been an attempt to account for these issues in the estimates of insect-induced stand mortality in Canada (Kurz and Apps, 1999). Insects responsible for major infestations in Alaska include the spruce beetle, the spear-marked black moth, the large-aspen tortrix, the eastern spruce budworm, and the larch sawfly.

Insect-induced stand mortality in Canada has been generally less than 1 million ha annually since about 1920, but has been greater than 2 million ha annually
during the 1940s and greater than 3 million ha annually during the 1970-80s (Kurz and Apps, 1999). The high levels of mortality during the 1940s and the 1970-80s were largely associated with the outbreak of the eastern spruce budworm in eastern Canada, which is hypothesized to occur on approximately a 30-year cycle. Insect infestations in Alaska peaked during the late 1970s (over 600,000 ha annually) with outbreaks of the spear-marked black moth, the large aspen tortrix, and the larch beetle, and during the 1990s (approximately 500,000 ha annually) with outbreaks of the spruce beetle, the eastern spruce budworm, and the larch sawfly (Werner et al., 2003). The outbreak of the larch sawfly during the late 1990s is estimated to have killed most of the larch in interior Alaska (> 650,000 ha; Werner et al., 2003).

Controls over the population dynamics of insects responsible for tree mortality are not well understood. Some insects seem to cycle on a regular basis, like the spruce budworm in eastern Canada (~30 years; Kurz and Apps, 1999), and the large aspen tortrix (~12 years) and the spear-marked black moth (~10 years) in Alaska (Werner et al., 2003). In comparison to infestations in eastern Canada, outbreaks of the eastern spruce budworm in Alaska have only been observed to occur in the 1990s when it infested around 600,000 ha (Werner et al., 2003). It has been hypothesized by Werner et al. (2003) that this infestation may be due to temperature-induced drought stress that has been experienced by white spruce in interior Alaska in association with warmer summers (see Barber et al., 2000). Factors that stress trees appear to be an important factor influencing insect outbreaks in Alaska (Werner et al., 2003). For example, *Ips engraver* beetles tend to infest stressed and dying trees near the edges of fires in interior Alaska (Werner and Post, 1985). Effects of warming temperatures on the life cycles of insects may also play an important role in insect outbreaks. For example, a single generation of spruce beetle can require either one or two years to mature, and this generally depends on temperature (Werner and Holsten, 1985). It appears that warmer summers in south-central Alaska during the 1990s may have caused a shift from a two-year to a one-year life cycle for spruce beetles that played a role in the large outbreak of the 1990s (Werner et al., 2003). Thus, the evidence from interior Alaska and elsewhere suggests that irruptive responses of insects are influenced by complex interactions between abiotic factors, such as temperature, and biotic factors, such as host plant physiology (Werner et al., 2003).

2.3 HUMAN LAND USE

Major uses of land that have affected terrestrial ecosystems of Canada and Alaska include agriculture and forest harvest. Agriculture in Canada is estimated to have expanded from 10.9 million ha of cropland in 1860 (based on Ramankutty and Foley, 1999) to 28.7 million ha in 1920 and then to 45.6 million ha in 1990 (McCuaig and Manning, 1982; Statistics Canada 1992 and the series of similar publications for each province). While most of this land was derived from the conversion of grassland ecosystems in the prairie provinces of Canada, it is estimated that there was a net deforestation of 12.5 million ha between 1860 and 1992 (based on Ramankutty and Foley, 1999). Since 1950, Canadian forests have had a net gain of 3.0 million ha at the expense of agriculture. While this is a low proportion of the total forest base (<1%), it is important to recognize that most of the afforestation has occurred in eastern Canada and deforestation continues to occur in western Canada. Thus, cropland establishment
and abandonment is an important process that is likely having different impacts across Canada.

While agricultural development in Alaska represents a miniscule proportion of the land cover, interior Alaska and adjacent Canada experienced substantial disturbance at the turn of the 20th Century in association with gold exploration. The height of this activity was felt when more than 1000 stern-wheeled riverboats operated on the Yukon River and 12 sawmills operated out of Dawson City, Yukon Territory and produced approximately 12 million board feet of lumber annually (Wurtz and Gasbarro, 1996; Naske and Slotnick, 1987). It is estimated that by the 1930s much of the accessible timber along the Yukon and Tanana Rivers in Alaska was exhausted (Wurtz and Gasbarro, 1996).

Annual forest harvest in Canada has approximately doubled from ~0.5 million ha in 1970 to ~1 million ha in 1990. The combined effect of increased forest harvest and other disturbances has led to a decrease in the mean stand age of Canadian forests from 81.5 years in 1970 to 78.2 years in 1990 (Kurz and Apps, 1999). Forest harvest in the Tanana River Basin of interior Alaska since 1970 has been less than 500 ha per year, which is a very tiny proportion of the estimated 600,000 ha of productive forests on state lands of the river basin (Crimp et al., 1997). During the latter half of the 20th Century, timber harvest in Alaska has primarily been concentrated on old-growth Sitka spruce coastal forests of the Tongass National Forest and private lands in southeast Alaska, where annual forest harvest increased over six fold from 1952 to 1992 (Joyce et al., 2002).

Recent trends of forest harvest rates in Canada and Alaska are substantially influenced by economics of the global forest sector as much of the harvested wood is exported out of the region to markets in Asia and the conterminous United States. Concern over conservation issues and the collapse of Asian economies in the 1990s have had substantial impacts in decreasing forest harvest in Alaska and the U.S. Pacific northwest during the 1990s. For the Canadian forest sector, the collapse of the Asian economies was somewhat offset by increased demand from the United States in the wake of decreased forest harvest in the U.S. Pacific northwest. One coupled ecological-economic assessment projects that climate change may have a negative impact on forest harvest of Canada over the next century even though it estimates an increase in timber growing stocks (Perez-Garcia et al., 2002). This occurs because the increased growing stocks are predicted to lead to lower prices. Canada is a country that largely exports harvested wood, and lower prices are expected to cause timber producers to adjust their losses by harvesting 1% to 3% less timber by 2100 in comparison to harvest rates without climate change (Perez-Garcia et al., 2002). These decreased harvest rates are expect to cause economic welfare losses to forest sector Canada of between 2 and 14 billion 1993 $US compared with analyses that assume no climate change (Perez-Garcia et al., 2002).

2.4 VEGETATION DYNAMICS

Climate change in Canada and Alaska is currently influencing land cover through effects on vegetation dynamics that are not directly associated with stand-level disturbances like fire and forest harvest. Climate warming has lengthened the growing season in Alaska (Keyser et al., 2000) and western Canada (Serreze et al., 2000), a pattern that has also been inferred to occur across the boreal forest region in analyses
based on satellite remote sensing and modeling of freeze-thaw dynamics (Myneni et al., 1997; Zhou et al., 2001; Dye, 2002; Zhuang et al., 2003). Lake ice has also been observed to be melting earlier across much the northern hemisphere in recent decades (Magnuson et al., 2000). The primary mechanism responsible for longer growing seasons appears to be earlier snowmelt in spring (Dye, 2002), which causes soils to thaw earlier (Zhuang et al., 2003). The seasonal transition of the northern landscape from a frozen to a thawed condition represents the closest analog to a biospheric on-off switch that exists in nature, affecting ecological, hydrologic, and meteorological processes dramatically (Running et al., 1999). The duration of the frost-free period bounds the length of the growing season in coniferous species of the boreal forest, and the timing of spring thaw is strongly linked to the annual carbon balance in boreal ecosystems (Frolking et al., 1996).

Figure 1. Advance in Northern North America Thaw Day.

Microwave remote sensing techniques have proven useful for mapping seasonal freeze/thaw dynamics in boreal/arctic latitudes (Frolking et al., 1999; Kimball et al., 2001). Beginning with its launch aboard the QuikSCAT platform in June 1999, the SeaWinds scatterometer has provided a capability for monitoring changes from frozen to thawed states across much of Canada and Alaska. The ability of the scatterometer to allow day and night monitoring of the land surface independent of cloud cover, combined with its moderate spatial resolution (~25 km) and wide swath width enable daily observation of the spatial and temporal dynamics of freeze-thaw cycles in the circumpolar high latitudes. Passive microwave data from the Special Sensor Microwave Imager (SSM/I) radiometers have detected changes across Canada and Alaska in springtime thaw date from 1988 through 2001, and indicate that the thaw date has advanced about 0.88 days per year across the 14-year period (Figure 1). The spatial distribution of this rate of change is complex, with some regions showing a tendency toward earlier thaw and others showing a later thaw. Finally, the higher resolution Synthetic Aperture Radars (SARs) provide a capability to monitor spatial and temporal patterns in springtime thaw across smaller regions, and allow examination of the spatial heterogeneity in the thaw transition. Merging the landscape thaw maps
with land cover and topography maps elucidates the spatial and temporal complexity of freeze/thaw processes with respect to variable landcover, drainage, slope and aspect.

Observations of permafrost warming in both western Canada and Alaska are associated with warming climate over the last 30 years (Romanovsky and Osterkamp, 1997; Osterkamp and Romanovsky, 1999; Vitt et al., 2000), a change that is consistent with observations that snowmelt is occurring earlier and that soils and lake ice are thawing earlier. Analyses based on satellite data suggest that both production and vegetation carbon storage have generally been enhanced across the boreal forest in recent decades (Myneni et al., 1997, 2001; Randerson et al., 1999; Zhou et al., 2001). Although there is substantial evidence that warming can increase production in the boreal forest through longer growing seasons and through enhancing soil nitrogen availability to plants (McGuire et al., 2003b), production may not increase if other factors limit production. For example, warming has reduced the growth in old white spruce trees growing on south-facing aspects in interior Alaska because of drought stress (Barber et al., 2000). At tree line in Alaska, the growth of trees located in warm dry sites below the forest margin declined in response to recent warming, whereas the growth of trees located at tree line, particularly in wet regions, increased (Lloyd and Fastie, 2002). Thus, there appears to be substantial spatial variability in the response of white spruce growth to recent warming in Alaska, and studies conducted elsewhere on other species throughout the boreal forest suggest that growth responses of warming depends on interactions between temperature and the timing and amount of precipitation (Briffa et al., 1998; D'Arrigo and Jacoby, 1993; Jacoby et al., 1985; Jacoby and D'Arrigo, 1995).

Climate warming also has the potential to alter the position of the treeline. The replacement of tundra with boreal forest occurred in earlier warm periods of Holocene in western Canada (Spear, 1993). Over the last half century, tree line advances into tundra have been documented in Alaska (Cooper, 1986; Suarez et al., 1999; Lloyd et al., 2003a; Lloyd and Fastie, 2003) and Canada (Morin and Payette, 1984; Scott et al., 1987; Lavoie and Payette, 1994). A tree-ring study to reconstruct the response of tree line on the Seward Peninsula of Alaska to warming indicates that spruce trees located in upland tundra have established progressively farther from the forest limit since the 1880s (Lloyd et al., 2003a). This has led to a conversion of shrub tundra into low-density forest-tundra within a band extending approximately 10 km from the forest limit. Both fire and the melting of permafrost may play a role in this treeline expansion (Lloyd and Fastie, 2002; Lloyd et al., 2003a). While changes in treeline on the Seward Peninsula of Alaska have been documented in the field, these changes have not been detectable by remote sensing technologies because the changes are not captured at the radiometric and spectral resolution of satellite imagery at decadal and subdecadal time scales (Silapaswan et al., 2001). In contrast to tree line changes, changes in the canopy structure of tundra have been detectable through analysis of photographic and satellite imagery at sub-decadal to multi-decadal scales. Two such studies have documented that tundra ecosystems in Alaska are becoming shrubbier on the North Slope (Sturm et al., 2001) and on the Seward Peninsula (Silapaswan et al., 2001) over the last several decades, observations that are generally consistent with large-scale remote-sensing studies (Myneni et al., 1997; Zhou et al., 2001). These studies are consistent with modeling studies in which additional wood growth is supported by inorganic soil nitrogen that is enhanced by summer warming (McKane et al., 1997; Clein et al., 2000; Le Dizes et al., 2003). Winter processes may
also play a role in supporting additional wood growth through both biophysical and biogeochemical effects of warmer and wetter winters (Sturm et al., 2003). Modeling studies suggest that the biomass of tundra ecosystems in Canada and Alaska is likely to increase with climate warming over the next century (McGuire et al., 2000; Le Dizes et al., 2003).

2.5 CHANGES IN THE AREA OF SURFACE WATERS

While there are observations that the melting of permafrost is leading the expansion of open water in the Tanana Flats of interior Alaska (Osterkamp et al., 2000), analyses of remotely sensed imagery generally indicate that there has been a significant loss of open water bodies over the past 20 years in many areas of Alaska and adjacent Canada. Significant water body losses have occurred in areas dominated by permafrost (Hawkins, 1997; Hawkins and Maltam, 2000), areas of discontinuous permafrost (Hinzman et al., 2001; Yoshikawa and Hinzman, 2003), and subarctic areas that are permafrost-free (Berg, 1999). The reduction of open water bodies may reflect a warmer and effectively drier climate, which is consistent with tree ring analyses that have shown temperature-induced drought stress in some areas of Alaska (Barber et al., 2000; Lloyd and Fastie, 2002).

One possible mechanism for the decrease in open water is an increase in evapotranspiration as temperatures rise and precipitation does not significantly increase. For example, Rouse (1998) estimated that under a 2 X CO$_2$ climate warming scenario, an increase in precipitation of at least 20 percent would be needed to maintain the present-day water balance of a subarctic fen. Lafleur (1993) estimated that a summer temperature increase of 4 °C would require an increase in summer precipitation of 25% to maintain present water balance. Another possible mechanism is associated with permafrost, which generally restricts infiltration of surface water to the sub-surface groundwater. However, unfrozen zones called taliks may be found under lakes because of the ability of water to store and vertically transfer heat energy. As climate warming occurs, these talik regions can expand and provide lateral subsurface drainage to stream channels (Figure 2). This mechanism may be important in areas that have discontinuous permafrost such as the boreal forest region of Alaska.
3 Consequences of Land Cover Change for the Climate System

3.1 RADIATIVE FEEDBACKS TO THE CLIMATE SYSTEM

Because most of the energy that heats Earth's atmosphere is first absorbed by the Earth's surface before being transferred to the atmosphere, the energy exchange properties of the land surface have a strong direct influence on climate. Tundra and boreal forest differ from more southerly biomes in having a long period of snow cover, when white surfaces might be expected to reflect incoming radiation (high albedo) and therefore absorb less energy for transfer to the atmosphere. However, winter albedo of tundra (0.6 - 0.8) is much higher than winter albedo of boreal forest stands, which varies between 0.11 (conifer stands) and 0.21 (deciduous stands) (Betts and Ball, 1997; Sellers et al., 1997; Hall et al., 1999; Hall et al., 2001), because short-statured tundra is snow covered while snow in boreal forest stands is substantially masked because of the canopy overstory of trees. The incorporation of better estimates of winter albedo for boreal forest stands into climate models led to substantial improvements in medium-range weather forecasting and in climate re-analyses (Viterbo and Betts, 1999).

During summer, albedo of boreal vegetation is lower than in winter, with deciduous stands and boreal non-forested wetlands having approximately twice the albedo of conifer forests (Chapin et al., 2000a; Chambers et al., 2003). This difference in albedo leads to fluxes of sensible heat (i.e., heat that can be sensed) in conifer stands that are 2 to 3 times those of deciduous stands, whereas the latent energy fluxes (i.e., evapotranspiration) of deciduous forest stands in the boreal forest are 1.5 to 1.8 times greater than those of conifer forest stands (Baldocchi et al., 2000; Chapin et al., 2000a). Because transpiration is tightly linked to photosynthesis, latent heat exchange tends to be dominated by transpiration in boreal forest stands with high productivity (e.g., deciduous forests). In contrast, evaporation plays a more important role than transpiration in the latent energy exchange of forest stands with low productivity (e.g., black spruce forests), where surface evaporation from mosses can account for up to half of total evaporation (Baldocchi et al., 2000). The substantial sensible heating over conifer stands leads to thermal convection and may contribute to the frequency of thunderstorms and lightning (Dissing and Verbyla, 2003), which plays an important role in the fire regime of the boreal forest as a source of ignition.

Because there are substantial seasonal and spatial differences in sensible and latent energy exchange in the boreal forest, climate warming has the potential to affect regional climate by altering both positive and negative feedbacks. Positive feedbacks associated with climate warming may result from lengthening of the growing season, from tundra that becomes more shrubby, and from the replacement of tundra with boreal forest. A longer growing season that leads to earlier snowmelt and later snow cover effectively reduces annual albedo and should lead to substantial heating of the atmosphere, which may further lengthen the growing season.

Expansions of shrub tundra into regions now occupied by sedge tundra, and of boreal forest into regions now occupied by tundra, would reduce growing season albedo and increase spring energy absorption and may enhance atmospheric warming (Bonan et al., 1992; Thomas and Rowntree, 1992; Foley et al., 1994; McFadden et al., 1998; Chapin et al., 2000a, 2000b). Regional modeling studies focused on Alaska have shown that the expansion of shrub tundra at the expense of sedge tundra may result in substantially warmer summers over tundra, with warming effects that extend into the
boreal forest of Alaska (Lynch et al., 1999; Chapin et al., 2000b). While these modeling studies clearly highlight that vegetation change in tundra has the potential to influence climate, the magnitude and extent of impacts on climate will depend on the temporal and spatial patterns of land cover change in the circumpolar Arctic.

Another positive feedback is associated with expansion of the boreal forest into regions now occupied by tundra. This would decrease albedo in both summer and winter and should cause substantial heating of the atmosphere, a response that could possibly accelerate the replacement of tundra by boreal forest (Bonan et al., 1995; McFadden et al., 1998; Chapin et al., 2000a, 2000b; Beringer et al., 2003). Studies conducted with general circulation models indicate that the position of northern treeline has a substantial influence on global climate, with effects extending to the tropics (Bonan et al., 1992; Thomas and Rowntree, 1992; Foley et al., 1994). At Alaska's latitudinal treeline on the Seward Peninsula, spruce trees have established progressively farther from the forest limit since the 1880s and have infilled within existing stands (Lloyd et al., 2003a). This has led to a conversion of shrub tundra into low-density forest-tundra within a band extending approximately 10 km from the original forest limit. These forests have a lower albedo and greater sensible heat fluxes than the tundra they have replaced, causing a net warming effect on the atmosphere (Beringer et al., 2003).

An increase in fire frequency for the boreal forest of Canada and Alaska is likely if climate continues to warm (Flannigan and Van Wagner, 1991; Stocks et al., 1998; Flannigan and Wotton, 2001; Flannigan et al., 2001). Land-cover changes from wildfire have the potential to influence energy exchange between the surface of boreal forests and the atmosphere. Fire in boreal forests of Canada and Alaska often leads to post-fire deciduous stands that last for approximately fifty years before being successionaly replaced by conifer stands. In contrast to the positive feedback to climate warming caused by forest expansion into tundra, any increase in fire frequency caused by boreal warming could lead to negative feedback to warming if fire increases the proportion of deciduous stands on the landscape. Post-fire deciduous stands have a higher albedo (0.14) than do conifer stands (0.09) which they replace and therefore transfer less energy to the atmosphere (Chambers et al., 2003). Thus, the degree to which the response of vegetation dynamics to climate warming influences regional climate depends on the interaction of factors that may both enhance and mitigate warming.

3.2 TRACE GAS FEEDBACKS TO THE CLIMATE SYSTEM

Important trace-gas feedbacks to climate of the Canada and Alaska region include feedbacks associated with the exchange of carbon dioxide (CO$_2$) and methane (CH$_4$) with the atmosphere (Roulet, 2000). Changes in these fluxes could either enhance warming (positive feedbacks) or mitigate warming (negative feedbacks) (Smith and Shugart, 1993; McGuire and Hobbie, 1997; McGuire et al., 2000; Chapin et al., 2000a; Clein et al., 2002). Boreal forests contain approximately 27% of the world's vegetation carbon inventory and 28% of the world's soil carbon inventory (McGuire et al., 1997). Much of the soil carbon is highly labile and has accumulated simply because of cold and/or anaerobic soils conditions (Neff and Hooper, 2003; Weintraub and Schimel, 2003). Recent high-latitude warming could trigger the release of this carbon that has accumulated over millennia.
Although the warming of aerobic soils will tend to increase the release of CO$_2$ from soils of Canada and Alaska, the net effect of warming depends on the balance between production and decomposition. The changing length of the growing season in Canada and Alaska may have important effects on production and carbon storage (Frolking et al., 1996). In temperate forests, annual carbon storage is enhanced by approximately 6 g C m$^{-2}$ for every day that the growing season is lengthened (Baldocchi et al., 2001), suggesting that longer growing seasons in boreal forest should also enhance terrestrial carbon storage (see also Frolking et al., 1996). In the boreal forest the start of the growing season, as defined by photosynthetic activity of the canopy, is tightly coupled to the thawing of the soil in conifer stands, because frozen soil prevents transpiration (Frolking et al., 1999; Zhuang et al., 2003). Deciduous stands begin net carbon uptake following leafout, which is also sensitive to the timing of snowmelt.

Figure 3. Cumulative changes in carbon stocks for Alaska from 1950 – 1995.

In situations where warming does enhance production of boreal forests, soil carbon storage will increase if the transfer of carbon from vegetation to the soil is greater than the enhancement in decomposition from warming. If this condition occurs, then the long-term rate of soil carbon storage depends on whether the carbon that is transferred to the soil decomposes quickly or slowly (Clein et al., 2000; Hobbie et al., 2000). Our understanding of soil carbon and nitrogen transformations in response to warming is incomplete and is a key gap that limits our ability to make projections of the long-term response of soil carbon to warming (Clein et al., 2000).

While fire activity is highly variable from year to year in specific regions of the boreal forest (Kurz and Apps, 1999; Stocks et al., 2000, 2002; Amiro et al., 2001; Conard et al., 2002; Kasischke et al., 2003), it has only recently been elucidated that fire in the boreal forest region as a whole has enough inter-annual variability that it may play a role in the inter-annual variability in the growth rate of atmospheric CO$_2$.
(Langenfelds et al., 2002). To understand how variability in regional scale fire activity influences carbon storage of the atmosphere, it is useful to examine the effects of fire on carbon storage at both the stand- and regional levels. Stand-level disturbances like fire are generally characterized by a period of ecosystem carbon loss, during which production is less than decomposition, followed by a period of ecosystem carbon gain once production exceeds decomposition (Kasischke et al., 1995; Zhuang et al., 2002). If regional disturbance regimes are at steady state for a substantial period of time, ecosystem carbon storage will not change because areas of carbon loss are balanced by areas of carbon gain. However, an increase or decrease in disturbance frequency will generally cause losses or gains, respectively, in carbon from terrestrial ecosystem to the atmosphere (Figure 3).

Together, both insect disturbance and fire have likely released substantial amounts of carbon into the atmosphere from Canada’s forests in the latter part of the 20th Century (Kurz and Apps, 1999; Chen et al., 2000; Amiro et al., 2001). An additional loss of carbon may occur if stands are salvage logged after fire. A large uncertainty in the loss of carbon in fire emissions of the North American boreal forest is the contribution from fire in forested peatlands, which might account for 10-20% of the area burned in western Canada (Turetsky et al., 2002). The burning of peatlands in Russia may have contributed to the large atmospheric carbon monoxide anomaly that was observed in 1998 (Kasischke and Bruhwiler, 2003; see also Langenfelds et al., 2002). Thus, the level of fuel consumption is an important factor in carbon dynamics of the North American boreal forest (Harden et al., 2000). Insect disturbance and forest harvest also result in carbon loss to the atmosphere, but the consequences of these disturbances on regional carbon dynamics is less clear. A major challenge is documenting the consequences of these disturbances for ecosystem processes and successional change through the cycle of disturbance and subsequent forest regrowth.

Whether changes in the distribution of vegetation result in carbon storage or loss largely depends on the type of transition (McGuire and Hobbie, 1997). For example, the replacement of tundra with boreal forest is likely to result in a net uptake of carbon from the atmosphere, whereas the transition of boreal forest to grassland is likely to result in a net release of carbon to the atmosphere. Equilibrium modeling studies suggest that the replacement of arctic tundra with boreal forest in Canada and Alaska has great potential to substantially increase vegetation carbon storage (gains of 14 to 18 Pg C among scenarios in McGuire and Hobbie, 1997), whereas soil carbon is likely to be less affected (loss of 1 Pg C to gain of 3 Pg C among scenarios in McGuire and Hobbie, 1997). Soil carbon was relatively insensitive in the transition from tundra to boreal forest in these simulations because increases in production were offset by increased rates of decomposition associated with increased soil temperature. This potential increase in ecosystem carbon storage by replacing tundra with boreal forest is likely to proceed at a very slow pace because the migration of boreal forests into tundra is likely to take several centuries (Starfield and Chapin, 1996; Chapin and Starfield, 1997; Lloyd et al., 2003a).

Although soil and lake drainage may be especially vulnerable to the response of permafrost to climatic warming, the net effect on radiative forcing of the atmosphere is not clear because drainage can either be enhanced or retarded by permafrost degradation, and because the response of drainage is likely to affect the release of CO₂ and CH₄ in opposite directions (Roulet et al., 1992a, 1992b; Roulet, 2000). For example, the release of CO₂ from aerobic decomposition is likely to be enhanced if
permafrost warming results in a drop of the water table (Oechel et al., 1995; Christensen et al., 1998), but emissions of CH$_4$ are likely to decrease because methanogenesis is an anaerobic process (Roulet et al., 1992a, 1992b; Roulet, 2000). In contrast, if permafrost melting results in the expansion of lakes and wetlands, then releases of CH$_4$ are likely to be enhanced (Reeburgh and Whalen, 1992; Zimov et al., 1997) with an associated reduction in CO$_2$ emissions. In eastern Canada peatlands, the enhanced CH$_4$ emissions associated with the creation of wetlands will likely result in a positive feedback to warming for up to 500 years until the enhanced storage of carbon in the wetlands (i.e., uptake of CO$_2$ from the atmosphere) offsets the enhanced radiative forcing associated with CH$_4$ emissions (Roulet, 2000).

3.3 FRESHWATER DELIVERY TO THE ARCTIC OCEAN

The delivery of freshwater from the pan-arctic land mass is of special importance because the Arctic Ocean contains only about 1% of the world’s ocean water, yet receives about 11% of world river runoff (Forman et al., 2000; Shiklomanov et al., 2000). The Arctic Ocean receives fresh water inputs from four of the fourteen largest river systems on earth (Forman et al., 2000). Additionally, the Arctic Ocean is the most river-influenced and land-locked of all oceans and is the only ocean with a contributing land area greater than its surface area (Ivanov, 1976; Vörösmarty et al., 2000). Freshwater inflow contributes as much as 10% to the upper 100 meters of the water column for the entire Arctic Ocean (Barry and Serreze, 2000). Changes in fresh-water inputs to the Arctic Ocean could alter salinity and sea ice formation, which may have consequences for the global climate system by affecting the strength of the North Atlantic Deep Water Formation (Aagaard and Carmack, 1989; Broecker, 1997). Modeling studies suggest that maintenance of the thermohaline circulation is sensitive to fresh-water inputs to the North Atlantic (Manabe and Stouffer, 1995). Also, freshwater on the Arctic continental shelf more readily forms sea ice in comparison to more saline water (Forman et al., 2000). The responses of freshwater inputs to the Arctic Ocean depend on changes in the amount and timing of precipitation, and the responses of permafrost dynamics, vegetation dynamics, and disturbance regimes to global change. For example, changes in evapotranspiration associated with permafrost and vegetation dynamics have consequences for river runoff that depend additionally on changes in precipitation inputs to terrestrial ecosystems.

The boreal forest plays a significant role in the hydrology of the circumpolar north because it dominates the land-mass that contributes to the delivery of freshwater to the Arctic Ocean. Over the past 70 years there has been a 7% increase in the delivery of freshwater from the major Russian rivers to the Arctic Ocean (Peterson et al., 2002; Serreze et al., 2003). In the Yenisey and Lena Rivers, this increase has occurred primarily in the cold season (Serreze et al., 2003). The pattern is most pronounced in the Yenisey, where runoff has increased sharply in the spring, decreased in the summer, but has increased for the year as a whole. While the mechanisms responsible for this pattern are not completely clear, the patterns are linked to higher air temperatures, increased winter precipitation, and strong summer drying. It is possible that the changes in runoff patterns for the Yenisey and Lena are associated with changes in active layer thickness and the thawing of permafrost (Romanovsky et al., 2000; Serreze et al., 2003). Fire disturbance is a factor that can accelerate the melting of permafrost (Zhuang et al., 2002), and may play a role in changes of runoff.
Although runoff has not been observed to increase in arctic and subarctic North American rivers like the Mackenzie and Yukon Rivers, which drain warming permafrost areas that have experienced increased fire frequency in recent decades, the ability to detect a trend for increased runoff may be limited by the shorter length of runoff records for these rivers (Peterson et al., 2002). A major challenge will be to understand the role of land cover change in the increased runoff of major Russian Rivers, and whether runoff of major North American rivers that drain into the Arctic Ocean is or is not changing.

4 Future Challenges

The quality of observations of land cover change in Canada and Alaska are in many ways very good in comparison to other high latitude areas. The history of fire since the middle of the 20th Century has been synthesized into spatial data sets that are useful for understanding the temporal and spatial patterns and controls of fire in Canada and Alaska. While data sets on the timing and extent of fire are operationally updated in the region, these updates are primarily derived from post-fire surveys of the perimeter of the burn scars, and satellite-based monitoring is not being used for this purpose (see Chapter 19). Satellite remote sensing could substantially improve these data sets in two ways. First, the area encompassed by the burn scar perimeter may result in an overestimate of the area burned if unburned inclusions are not taken into account. Satellite remote sensing could contribute substantially by estimating the area of these inclusions. Second, burn severity (i.e., biomass consumption) is important for understanding the impacts of fire on changes in carbon storage, particularly in soils. Satellite remote sensing could contribute substantially to providing spatial information on burn severity (Michalek et al., 2000; Isaev et al., 2002).

In contrast to fire, our understanding of other disturbance regimes such as insect disturbance, forest harvest, and changes in croplands primarily comes from agency-based statistics, many of which are at state and provincial levels. While operational remote sensing of these disturbances is not currently a reality, the development of this capability has the potential to provide data sets that would allow a more comprehensive understanding of the temporal and spatial patterns and controls of these disturbances. Remote sensing has contributed substantially to our understanding of changes in the length of the growing season in Canada and Alaska, and remote sensing case studies have informed us that the structure of tundra vegetation and the area of surface waters are changing in Alaska. While we have ground-based data indicating that tree line is changing in both Canada and Alaska, the detection of these changes with satellite imagery remains a substantial technical challenge for the remote sensing community (Masek, 2001).

The synthesis of observations on land cover change in Canada and Alaska has allowed us to estimate some of the consequences of these changes over the last century, particularly the impacts on carbon storage of the region. While there have certainly been carbon losses associated with an increase in fire, insect, and forest harvest in the last half of the 20th Century, there are uncertainties about how these compare to mechanisms that might increase carbon storage. These mechanisms include climate warming, nitrogen deposition, and increases in atmospheric CO2. Inventory and process-based approaches disagree as to whether the region has been a small source or
sink for atmospheric CO$_2$ of around 0.1 Pg C per year in recent decades (Kurz and Apps, 1999; Chen et al., 2000; Yarie and Billings, 2002).

An alternative to these approaches is the atmospheric inversion approach, which estimates the timing and location of sources and sinks based on measurements of atmospheric CO$_2$ at around 100 stations around the globe. Only in the last few years have inversion techniques become advanced enough to evaluate carbon fluxes over subcontinental regions and to estimate inter-annual variability in these fluxes. While some atmospheric inversions suggest that the region was a sink of 0.2 to 0.4 Pg C per year during the 1980s, Gurney et al. (2002) in an analysis that used 16 different transport models suggests that the North American boreal region was a source of approximately 0.2 Pg C per year between 1992 and 1996. More recently, Rodenbeck et al. (2003) performed an inversion covering the period 1983 to 1997, and, consistent with Rayner et al. (1999), Dargaville et al., (2002a) and Gurney et al. (2002), estimated a sink of increasing strength during the 1980s and into the early 1990s, which peaked around 1993 and then rapidly changed to being a source during the mid-to-late 1990's. Thus, analyses based on atmospheric inversions suggest that carbon exchange of the North America boreal forest is very dynamic.

Figure 4. Cumulative changes in carbon stocks for Canada from 1950 – 1995.

Simulation models that consider changes in atmospheric CO$_2$, climate, and fire suggest that responses to atmospheric CO$_2$ and climate in the region might more than compensate for carbon losses associated with land-cover change (Figure 4). However, such an analysis for the region is incomplete because it does not consider all the factors that might influence carbon storage. There are substantial uncertainties in estimating regional changes in carbon storage with both process-based and inversion approaches (McGuire et al., 2001; Prentice et al., 2001; Schimel et al., 2001; Dargaville et al., 2002a, 2002b; Gurney et al., 2002). Better information on the timing, extent, and severity of disturbance would decrease uncertainties in process-based approaches that require disturbance information to properly assess the effects of land cover change on carbon storage in the region.

Better information on the timing, extent, and severity of disturbance and land-cover changes is important for better understanding controls over land cover change in
Canada and Alaska and for building a capability to predict future changes in land cover in the region. This capability is important to develop because the region represents about 10% of the vegetated cover of the earth and changes in land cover could have substantial impacts that influence the climate system through radiative, trace gas, and Arctic Ocean freshwater feedbacks to the climate system. There is substantial evidence that changes in land cover over the last century are associated with both climate change and human influences. Preliminary analyses indicate that climate change and human activity have the potential to interact in ways that may have substantial influences in the regional and global forest sectors in the future (Perez-Garcia et al., 2002). Progress in representing how both climate change and humans influence the land cover of Canada and Alaska in predictive analyses of global change poses a major ongoing challenge. It is clear that continued development of data sets on land cover change will substantially contribute to this capability.

5 REFERENCES


LAND COVER DISTURBANCES AND CLIMATE


CHAPTER 10

MAPPING DESERTIFICATION IN SOUTHERN AFRICA

STEPHEN D. PRINCE

Geography Department, University of Maryland, Room 2181 LeFrak Hall, College Park, MD 20742-8225 USA

1 Introduction

Land degradation over vast areas in tropical semi-arid regions, as a result of persistent drought and inappropriate land management, was brought to international public attention in the early 1980s following two catastrophic droughts in the African Sahel within a 10 year period. It was ultimately concluded that an interaction between climate and human land use was occurring at a sub-continental scale and that new techniques were needed to monitor and explain these processes. Similar situations have been recognized throughout the world, some in quite different climates (e.g., Mabbutt and Floret, 1980). While there has been significant progress in the study of the land-atmosphere interactions involved in desertification (Xue and Fennessy, 2002), studies of the ecological components of the process of desertification have mainly been at the local scale related to, for example, wind, gully and sheet erosion, bush encroachment, and salinization. The land surface processes of desertification also operate over large areas and long time scales (Prince 2002), and these have not received much attention. This is mainly because of the difficulty in obtaining appropriate data, although unfamiliarity with these time and space scales may also have contributed to the lack of progress. Studies of the anthropogenic component of desertification have largely assumed the biophysical nature of desertification, and have dealt with socioeconomic (Vogel and Smith, 2002), governmental and political perspectives (Chasek and Corell, 2002) assuming that desertified areas have the same properties as arid ecosystems.

Following a discussion of the significance of time and space scales in the definition of desertification, this chapter explores the potential of remote sensing of annual primary production to detect desertification. The concept of reduction in mean, interannual, potential production is applied to the country of Zimbabwe, which has several features that make it suitable for the assessment of the extent of actual desertification as defined here. Finally the results are discussed in the context of global monitoring of desertification.

2 The Nature of Desertification

Certain processes, distinct from drought and operating over periods of more than a single year, may be occurring in degraded, semi arid lands. These processes are hard to detect owing to the extreme inter-annual and spatial fluctuations in the environment of regions subject to drought and over-utilization (Lebel, Taupin et al., 1997). Degradation of the type that led to the international concern in the 1980s occurs in a diffuse pattern over very large areas. Within these areas some or all of the well-known processes of
erosion, bush encroachment and salinization may be found in different intensities. Regional processes in addition to local causes seem to be operating and regional mechanisms therefore need to be identified. These mechanisms might include the mesoscale climate systems that lead to persistent drought (Le Barbe and Lebel, 1997), the transport of materials by wind and water, and the areas over which human utilization acts as a result of the socio-economic systems and government policies in these regions. For example, in semi arid regions, nomadic and transhumant societies have often arisen that exploit the large regional and temporal variability in rainfall. However, over the past 50 years, most governments have attempted to sedentarize these populations thus changing land use of entire regions, often \( >10^6 \) ha.

Another, equally important yet less well-publicized body of theory has emerged in terrestrial plant community ecology and rangeland management, that of multiple stable or semi-stable states. This concept suggests that less productive land conditions might become established that are very difficult to reverse (Weber, Moloney et al., 2000; Jackson and Bartolome, 2002; Walker, Abel et al., 2002). Clearly the implications for management of a system that will not return to its former productivity once rain returns or utilization is reduced are very different from those of one that will.

Prince (2002) has suggested that the term desertification be reserved for processes distinct from those responsible for local and short-term degradation. The characteristic time and length scales of these processes were tentatively set at 20-50yr and 50-100km, but these require further study. The spatial scale of the ecological changes that characterize desertification might arise from several processes. While there is a case for treating desertification as a distinct phenomenon, different from the well-known, short-term effects of drought and local scale land degradation, suitable tools to measure and explore any process that occurs at multi-year and large area scales are not yet widely available. The need for mapping of desertification has been evident at least since the 1974 drought in the Sahel and the subsequent 1977 United Nations Conference on Desertification (UNCOD), organized by the United Nations Sahelian Office (UNSO, renamed the UNDP Office to Combat Desertification and Drought in 1994) and the United Nations Environment Programme (UNEP). Various attempts have been made to inventory desertification in order to assess the magnitude of the problem and to provide a baseline for monitoring (e.g., Mabbutt and Floret, 1980; Oldeman, Hakkeling et al., 1990; Middleton, Thomas et al., 1997; CSIR 1999), but the lack of any readily measured, objective indicators of actual degradation applicable at a regional scale has inhibited progress.

3 Existing Assessments of Soil Degradation

The current World Atlas of Desertification (Middleton, Thomas et al., 1997) is based on the Global Assessment of Soil Degradation (GLASOD), undertaken by the International Soil Reference Center (ISRIC) (Oldeman et al., 1990). GLASOD was not developed specifically for desertification studies, but to map human-induced soil degradation in all environments and it included an assessment of the human activities responsible for degradation. GLASOD was methodologically qualitative, based on opinions of a large number of regional soil degradation experts (Thomas and Middleton, 1994) in a “structured informed opinion analysis” adopted from (Dregne
The assessments were made using a pre-defined set of criteria for each of the four main degradation processes: water erosion; wind erosion; physical deterioration; and chemical deterioration of the land. For each mapping unit, the degree of soil degradation caused by these four processes was classified as either light, moderate, strong, extreme, or no degradation. Five categories related the extent of the degradation from infrequent to dominant. A matrix combining the degree and extent of degradation was then used to give the overall severity for a given mapping unit. The map appears to be a quantitative assessment of desertification, though in reality it is a quantification of qualitative assessments. At a scale of 1:15,000,000, GLASOD is only usable for areas >1°x1°.

(Eswaran, Reich et al., 2003) have mapped global vulnerability to desertification using the FAO/UNESCO Soil Map of the World (FAO 1992) at a scale of 1:5,000,000 (interpretable to approximately 0.5°), whose units were converted to taxa of soil taxonomy. In addition, a climate database with records for about 25,000 stations globally was used to obtain soil moisture and temperature regimes. The soil and pedoclimate information was used to place each map unit into one of nine land quality classes. To facilitate placement into these classes, a list of 24 land stresses that constrain grain production was developed. An assessment of vulnerability to desertification was then made using the procedure of (Eswaran and Reich, 1998). The map shows vulnerability to, not actual, desertification and is another type of expert interpretation, hence it is not appropriate for monitoring.

Many maps of desertification have been prepared for individual countries and smaller areas but, with no agreement on the criteria for the designation of desertification, these have limited use for comparison between regions. For example, a similar approach to GLASOD has been pursued by (Hoffman and Todd, 2000) in the Republic of South Africa, but at a finer scale. In Zimbabwe erosion features were mapped by (Whitlow 1988). The emphasis on the human perception of degradation in these studies is their strength. However, in most cases, the nature of the data used does not allow changes in land condition to be detected, which severely limits the value of the techniques for monitoring.

### 4 Measurement of Desertification by Reduction in Primary Production

Quantitative measurement of desertification has been an elusive goal, partly because of the uncertain definition of desertification, which limits mapping using variables that have an explicit relationship with the processes of interest (Prince, Justice et al., 1990; Rhodes 1991). Central to the definition of desertification adopted by the UN is the concept of reduced productive potential of the land (UNCOD 1977) and the role of the carbon cycle, e.g., “the spread of desert-like conditions of low biological productivity due to human impact under climate variations” (Reynolds 2001). The aspect of productivity most relevant to desertification is net primary production (NPP), that is, the accumulation of biomass through time. On this foundation it is hypothesized that desertification reduces NPP below the undesertified state. Other changes in the carbon cycle follow from this primary effect, such as reduced biomass and soil carbon, which may or may not be present, depending on the exact nature of the desertification.
Various metrics of the interannual dynamics of regional scale NPP have been suggested as appropriate methods for measuring desertification (Prince, Brown de Colstoun et al., 1998; Prince 2002). Desertification has an important human dimension owing not just to the contribution of inappropriate land utilization in causing desertification, but also to the need to judge the state of the land relative to the potential supply of ecosystem goods and services under existing land uses. Thus NPP alone is not adequate to measure desertification, rather it is the productivity of the land relative to its undesertified state that is of interest. This potential NPP could be measured for limited regions in large, long-term field study sites, but this is unlikely to be practical for countrywide applications (Pickup 1998). Remote measurement using satellite observations offers a more immediate solution.

Techniques for measurement of NPP using satellite data were first developed in the mid 1980s (Prince and Justice, 1991) but it is only now that an archive has accumulated with a long enough record (20+yr) to allow desertification studies at appropriate time scales (Prince, Goward et al., 2000). Procedures for using satellite data to map desertification, based on monitoring the actual and potential NPP of the land, have been developed (Prince, Brown de Colstoun et al., 1998). The premise is that reductions of the NPP below that of equivalent, non-desertified land provide an indicator of desertification. Three approaches to measurement of desertification using reduced NPP have been proposed.

4.1 RAIN USE EFFICIENCY (RUE)

Rain Use Efficiency (RUE) is the ratio of NPP to rainfall (Prince, Brown de Colstoun et al., 1998). NPP has been shown to be directly related to rainfall in many studies in semi-arid regions (Lamotte and Bourlière, 1983; Le Houérou 1989) hence rainfall may be used as a surrogate for potential production. Thus the RUE ratio measures the actual production as a function of potential production, and low RUE values provide a useful index of degradation, independent of the rainfall. RUE is based on a simple concept that is confirmed in the many comparisons of NPP with rainfall (Le Houérou 1984, 1988; Noy-Meir 1985), however it is only applicable where rainfall is the principal limiting factor of potential NPP which is only true in drier areas (<1000mm in the tropics). In higher rainfall areas this is not the case, and even in lower rainfall areas, factors other than rainfall can determine local NPP. (Prince, Brown de Colstoun et al., 1998) discuss the use of soil moisture for this purpose.

4.2 POTENTIAL-ACTUAL NPP (PAN)

A second, more mechanistic technique using a biogeochemical model to estimate potential NPP has been developed (Prince 2002). This method takes into account soils, nutrients, and soil moisture. The index of desertification in this case is the difference between potential and actual NPP, where a comprehensive suite of conditions in addition to rainfall is used to estimate the potential NPP. The technique is not limited to the drier parts of the world where potential NPP is well correlated with rainfall. It also has the advantage of being measured in units of NPP, so PAN is a direct measure of the impact of desertification on productivity.
Both RUE and PAN are subject to the practical limitation on spatial resolution caused, not by the satellite data, but the availability of rainfall data and, for the PAN technique, also by the coarse resolution of most available soil maps. Meteorological stations are sparsely distributed and local rainfall can be spatially highly variable in semi-arid regions (Lebel, Taupin et al., 1997). Thus, if 0.5° x 0.5° gridded rainfall data are used (the finest resolution practical at a global scale), the spatial resolution of the desertification map is limited to >50km, a scale that makes identification of local causal factors quite difficult. Advances in remote sensing of rainfall are being made and it is anticipated that finer temporal and spatial resolution measurements may soon be available (Flaming 2002).

4.3 LOCALLY SCALED NPP (LNS)

To address the problem of sparse surface data, a third technique has been developed, Local NPP Scaling (LNS), in which the NPP of each pixel is expressed as a proportion of the maximum observed in all land falling into the same land type (Pickup 1996). Stratification by terrain type allows climate, soil and land cover differences to be normalized, and desertification to be detected relative to the maximum observed NPP. In fact cultural factors could be included, such as commercial or communal land tenure.

It is hypothesized that reduced NPP, below the potential set by the biogeophysical conditions, will highlight potentially desertified areas, and this third approach is tested here. Several uncertainties exist, for example, it is not clear whether the LNS technique will be able to estimate potential production. Other causes of differences in scaled NPP are also possible, such as inability to find undegraded land in a stratum or a potential production classification that wrongly includes some areas that have a higher potential NPP from another class.

4.4 TEMPORAL ASPECTS OF THE DETECTION OF DESERTIFICATION

It has been noted by (Pickup and Chewings, 1994) that it is in wet periods that desertification can best be detected. During wet periods reductions in NPP owing to desertification are separable from reductions that are caused simply by drought. NPP is measured for entire growing seasons, using multitemporal satellite observations; individual observations of the vegetation do not capture the time component of carbon sequestration. Degraded land often supports brief flushes of ephemeral weeds that can make the identification of degradation difficult using a single satellite observation (Fabricius 2002). A potential problem with RUE, PAN and LNS, however, lies in the non-equilibrium behavior of semi-arid lands, where trends are masked by short-term temporal and spatial variation in rainfall (Pickup, Bastin et al. 1998). Long enough periods of time and large enough areas must be included to allow the natural lags in vegetation recovery from rainfall and other perturbations (Goward and Prince, 1995). Further studies are needed to establish the appropriate time period.
5 ZIMBABWE – A Case Study

5.1 INTRODUCTION

The validation of the techniques outlined in Section 4 requires independent data on desertification, mapped at a regional scale. Apart from the GLASOD map (Oldeman, Hakkeling et al., 1990) and USDA Natural Resource Conservation Service (NRCS) Desertification Vulnerability map (Eswaran and Reich, 2003) (Figure 1), which are rather too coarse, these data are hard to find. Some regions of southern Africa seem to be in danger of desertification, as defined in Section 2, and the case of Zimbabwe is particularly interesting because of the strong contrasts in sustainability of land use practices (Gore, Katerere et al., 1992) and the existence of a comprehensive map of the extent of erosion (Whitlow 1988).

![Fig.1. Extracts from existing global degradation and desertification maps for Zimbabwe and parts of surrounding countries. Original colors converted to grayscale.](image)

- Global Assessment of Soil Degradation (GLASOD) map (Oldeman et al., 1990).

- Human Induced Desertification Risk map. Based on an overlay of the NRCS global desertification map and a global population density map (Eswaran and Reich 2003).
Land has been a source of political conflict in Zimbabwe since colonization, both within indigenous black communities and especially between white settlers and the black rural communities. Under British colonial rule and under the white minority government that unilaterally declared its independence from Britain in 1965, white Rhodesians took control of the vast majority of good agricultural land, leaving black peasants with marginal “tribal reserves” (Roder 1964; Moyo, Robinson et al., 1991). Over the years, a large agriculture and livestock industry has been developed on the better land, now known as commercial land, while the rural indigenous population is confined to so-called communal lands (Figure 2).

The government of Zimbabwe has recently begun to address the land issue by means of a so-called “fast track” land redistribution program (Moyo, Rutherford et al., 2000; Human Rights Watch 2002). This policy has thrown the country into a state of ferment (Bush and Szeftel, 2000; Chattopadhyay 2000), which has reduced commercial production, pushed up the price of staple foods and exacerbated inflation. Although much of the public attention has concentrated on illegal land occupations of the commercial lands and the democratic crisis, the vulnerability of the large population of the degraded communal lands is the most serious humanitarian issue. On top of this, a severe drought in the 2001/2 wet season reduced rain-fed production (USAID 2003). Communal land residents are subject to at least three sources of vulnerability; the chronic unsustainability of production on the land they occupy; the rising price of food caused by the loss of commercial production; and the drought. Until 2002 Zimbabwe’s commercial sector was sufficiently productive to be the principal source of grain for

---

**Fig. 2.** Maps of land allocation (administrative units, FEWS 1996) and population distribution in Zimbabwe. Note the correspondence of the high population and communal areas.

*a* Fourth level administrative units and land use, Zimbabwe (FEWS/ARD 1996).

*b* Zimbabwe rural population density 1982 (Davies and Wheeler 1984).
much of central southern Africa. The loss of commercial production in Zimbabwe has endangered food security of the entire region.

The condition of the land in the communal areas has caused concern for many years (e.g., Hamilton 1965; Whitlow and Zinyama, 1988; Zinyama 1988; Stone and Dalal-Clayton, 1992), and seems to be getting worse. Significant parts of the land area of Zimbabwe are seriously degraded and less productive than they need to be to support their current population (Moyo, Rutherford et al., 2000; Lado 1999; Whitlow 1988; Zinyama and Whitlow, 1986; Kay 1975). The processes leading to land degradation in Zimbabwe involve the interaction of political, social, economic, and environmental factors (Gore, Katerere et al., 1992). Most of this degradation has occurred in the communal lands, where population pressures are greater and resource management practices weaker than they were in the intensive commercial farming areas before fast-track land redistribution started, or in the general lands (Prince 2000). Historically the transportation infrastructure was developed to serve the commercial lands and bypasses most communal land, moreover, land tenure systems in the communal lands are not conducive to small-scale commercial production even if markets were accessible. Some of the degradation seems to date back to the civil war period in the 1960's, when conservation efforts were neglected due to security problems.

A comprehensive baseline of degradation in the early 1980s is provided by a national erosion survey (Whitlow 1988, 1989). The survey used aerial photography flown mainly between 1981 and 1984. Rates of soil formation in the semi-arid tropics are very slow (e.g., 400 kg ha$^{-1}$yr$^{-1}$), whereas rates of soil erosion are very much greater; estimates for average soil losses on crop lands and grazing areas on commercial farms are 15,000 and 3,000 kg ha$^{-1}$yr$^{-1}$ respectively; the equivalent average for communal lands were 50,000 and 75,000 kg ha$^{-1}$yr$^{-1}$ (Whitlow 1988). These rates are averages and more recent observations indicate that they may need revision (Environmental Software and Services 2002).

There is a direct and positive correlation between increases in the extent of eroded terrain, soil type and increases in population density (Zinyama 1988). This relationship is especially strong in the communal lands, which are located, more often than commercial land, in the poorer agro-ecological regions (regions III, IV and V). Thus the role of land use practices in degradation is hard to separate from the inherent properties of the land. Nevertheless, over utilization has occurred even on better lands in communal areas (Prince, Reshef et al., 2002). The only areas where negligible soil erosion was observed in the early 1980s were national parks and wildlife areas. The consequences of the erosion described by Whitlow are seen in general declines in crop yields. In some areas the cultivation of maize may cease to be possible as soils become too shallow for crop growth, and sorghum cultivation may subsequently be affected (Whitlow 1988).

Since it was created using a completely independent technique, the baseline provided by the Whitlow survey (Whitlow 1988) provides a valuable comparison with the maps of RPPP presented here. Furthermore, the early 1980s were a significant turning point in the political history of Zimbabwe, when land use began to change more rapidly.
5.2 METHODS

5.2.1 Mapping degradation

The satellite data available for estimation of NPP for Zimbabwe are shown in Table 1. The extent of the country, and the spatial grain of its land use make a resolution of 1km$^2$ or finer desirable so that the impacts of variation in land potential and land use can be detected. Landsat and SPOT HRV data have even finer resolution, but the resulting volume of data are prohibitive, especially when the interest is in land surface dynamics extending over 20 years, moreover the temporal frequencies of data acquisition are not suited to the measurement of NPP (Prince and Tucker, 1986). MODIS data are only available for a short period so far. AVHRR HRPT data have the longest archive, extending back to 1985, however these data are not publicly available at this time. The AVHRR GAC data have too coarse a resolution for an individual country study.

<table>
<thead>
<tr>
<th>Sensor name</th>
<th>Available data archives</th>
<th>Net Primary Production</th>
</tr>
</thead>
<tbody>
<tr>
<td>SPOT HRV (20m, 26 days)</td>
<td>1986-present</td>
<td>Not available</td>
</tr>
<tr>
<td>Landsat TM/ETM+ (30m, 16 days)</td>
<td>Limited from 1982 to 1999, improved from June 1999-present</td>
<td>Not available</td>
</tr>
<tr>
<td>MODIS (1, 0.5, 0.25km, 1-2 days)</td>
<td>EOSDIS Starts Nov 2000-present</td>
<td>Starts Dec 2000</td>
</tr>
<tr>
<td>SPOT VEGETATION (1km, 1 day)</td>
<td>1998 March - present</td>
<td>Not available</td>
</tr>
<tr>
<td>AVHRR LAC/HRPT (1km, 4-9 days)</td>
<td>Satellite Applications Centre, RSA 1985-present</td>
<td>In preparation (Wessels, Prince et al. 2003)</td>
</tr>
<tr>
<td>AVHRR GAC (8km, 4-9 days)</td>
<td>1981-2000 Pathfinder AVHRR Land (PAL)</td>
<td>1981-2000</td>
</tr>
</tbody>
</table>

Table 1. Satellite sensors suitable for estimation of net primary production in Zimbabwe.

SPOT VEGETATION data are used here. The data have appropriate time and space resolutions but data are available for only four growing seasons, and the spectral radiances that the instrument measures are not suitable for modeling of primary production, unless alternate sources of surface temperature, soil moisture and humidity are available with the same 1km$^2$ resolution. As a result this case study is only able to address degradation that manifests itself over a period of four years, an interval that does not qualify for the term desertification as defined in Section 2. The degradation that is detected will overestimate desertification, since areas that experience shorter term reductions in realized potential primary production may recover. Nevertheless the patterns that emerge are quite stable, suggesting that the detected degradation is not ephemeral.
The seasonal sum of the normalized difference vegetation index (NDVI) was used (Figure 3) as a surrogate for net primary production (NPP) (Prince 1991). Summed NDVI is a reasonable approximation of NPP in grassland, cropland and sparse woodland. More sophisticated modeling of NPP can be undertaken (Prince Goward 1995) but it was considered more important to estimate productivity at 1km² resolution rather than use NPP for which the highest multi-year resolution is 64km².

Fig.3. Mean potential vegetation productivity for Zimbabwe and bordering regions. Summed SPOT VEGETATION NDVI, May 1998-Jan 2002 (dark – high, light – low NDVI).

Two factors, for which maps are available at an adequate resolution, seem to account for most of the variability in productive potential in Zimbabwe, these are precipitation (Figure 4a) and land cover (Figure 4b). Therefore a map was created of potential production classes by intersecting these two data layers. The ten-day accumulated precipitation data were summed to create an annual precipitation map. Next, the summed precipitation layer was partitioned into three classes using natural breaks in the range of values. The land cover classes were aggregated into seven classes. The urban and bare ground pixels were aggregated into a single class. Finally the three precipitation classes and seven land cover classes were intersected forming a map of 21 separate potential production classes (Figure 4c).

The 10-day SPOT-VEGETATION NDVI composites were summed from May 1998 to Jan 2002. Using the potential production class map (Figure 4c), the NDVI sum values falling in each class were extracted and the 90th and 10th percentile NDVI sum values for each class calculated. Values ≥90th and ≤10th percentiles were assigned potential primary productivity indices of 100 and 0 respectively, where 0 represents land completely degraded and 100 land at its potential primary production. The values of this index were mapped to create a realized percentage of potential productivity (RPPP) map. This method assumes that all pixels within a class have the same productive potential and that each class has some undegraded areas.
5.2.2 Validation

The RPPP map was compared with independent maps of degraded areas. First, images of Landsat data were made for areas that had a wide range of RPPP values. The high spatial resolution of the Landsat (30m) allowed land cover, including sparsely vegetated areas, to be detected visually. Second, the RPPP map was compared with the National Land Degradation Survey of Zimbabwe (Whitlow 1988). The Whitlow study was based
on photography mainly in 1980, although some photographs were for as early as 1979
and others as late as 1984, approximately 20yr before the satellite data. Sustainability of
land use or the lack of it is a medium-term phenomenon, thus correspondence of
severely eroded areas in 1980 to low realized percentage of potential productivity in
2000 can be regarded both as a test of the skill of the approach and an indicator that the
observed degradation may, in fact, be a case of desertification.

5.3 RESULTS

5.3.1 Realized percentage of potential production (RPPP) map
The RPPP map (Figure 5) indicates severe degradation in large areas of Zimbabwe. Of
the 9,565 pixels sampled, 31% had a realized percentage of potential productivity value
equal to 0, and 53% ≤15 (mean 20, mode 13, standard deviation 24).

Fig. 5. Realized percentage of potential production (RPPP) for Zimbabwe. Based on local NDVI scaling (LNS) of SPOT VEGETATION NDVI data for May 1998-Jan 2002 and a potential
production classification (Fig. 4c).

Darker areas less degraded, lighter areas more degraded. Fourth level administrative boundaries included for interpretation.
The map indicates some very clear patterns in the distribution of degradation. Overall land in good condition, relative to its potential, occurs in a forward-leaning “U” shape, in a broad sweep across the north west of the country (see Figure 6 for place names) from Mhangura (Mashonaland West), south west to Lupane (Matabeleland North), south east through Bulawayo to Rutenga (Masvingo), then north east to Chimanimani (Manicaland) and north through Mutare along the Mozambique border to the Inyanga highlands. The “U” shape encloses the “highveld”, the most productive, large scale commercial croplands which, as is to be expected in cultivated lands, have a lower realized percentage of potential productivity. Outside the less degraded “U” shaped area, to the north, northwest, south, and north east less good conditions prevail and lower values are found. Thus the overall countrywide pattern is one of an agricultural core with average realized percentage of potential productivity, surrounded by land close to its potential production, and outside that again, land which is much more degraded.

The country-scale pattern is interrupted by land in striking contrast to the broad configuration outlined above. Careful comparison of the RPPP with the population and land tenure maps of the country (Figure 2) indicates a remarkable association of low indices, indicating severe degradation, with communal areas (Table 2). Some communal areas stand out as large areas of severe degradation. These include the Save catchment in Manicaland, an area centered on Mutoko (Mashonaland East), along the Shashe river in Matabeleland South, and in the north Midlands centered on Gokwe. Other large areas in equally poor condition occur, but the forgoing are the most obvious centers on a national scale.

Although communal areas are often located in land areas with intrinsically low potential, the RPPP has taken this effect into account by using a potential production class map (Figure 4c).

<table>
<thead>
<tr>
<th>Tenure type</th>
<th>Number of 1km² pixels</th>
<th>Percent pixels with RPPP ≤ 35</th>
<th>Mean RPPP</th>
<th>Median RPPP</th>
<th>Standard Deviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Communal</td>
<td>202,375</td>
<td>53.6%</td>
<td>39.6</td>
<td>32</td>
<td>36.3</td>
</tr>
<tr>
<td>Commercial</td>
<td>209,424</td>
<td>27.4%</td>
<td>59.8</td>
<td>58</td>
<td>36.4</td>
</tr>
</tbody>
</table>

Table 2. Realized percentage of potential primary production (RPPP) statistics for communal and commercial land in Zimbabwe.
and yet the severe degradation pattern remains. Interestingly, not all communal or commercial are identified as degraded or not degraded, respectively, as might be expected if only productivity were used.

There is a trend of increasing productivity in Zimbabwe from the southwest to the northeast (Figure 3). This is due to the increase in precipitation from the SW to NE of the country (Figure 4). Consequently, without the use of potential production classes and scaling, one might wrongly assume that the SW is more severely degraded than the NW region. This is not the case, since the ecological potential of the SW region is lower than that of the NE due to the precipitation limitation, and the RPPP identifies other areas as more degraded. The potential production class map is intended to remove this trend.

5.3.2 Validation of the realized percentage of potential production map
Zimbabwe has innumerable examples of stark contrasts in land condition across political and administrative boundaries, often separated only by a fence line. The Landsat data with their 30m resolution illustrate qualitatively the condition of the land at a local scale. An example of these contrasts is shown in Figure 7. It is very clear at this fine scale that low RPPP values are associated with sparse vegetation and communal land tenure. While not proof of low sustainability, this association of sparse vegetation and low RPPP provides circumstantial support for the validity of the map. The distinctive geometrical borders of the sparse vegetation, which persist even when viewed at high spatial resolution, suggest that low values of the RPPP are associated with land use. Commercial farms with high RPPP are surrounded by extremely degraded communal lands, which suggests that these communal lands are not degraded simply because they have low ecological potential. Furthermore some communal land has a high RPPP, and is therefore probably not degraded.

Fig. 7. a) Landsat 7 ETM+ high resolution images of Zvishavane (southern Midlands) showing areas of strongly contrasted primary productivity. The dark area is large-scale commercial land and the lighter surrounding area, communal lands. The A9 road crosses the image from SW to NE. Bands 3, 2, 1. WRS path 170 row 74, acquired Apr. 23rd 2000. b) The same part of the realized percentage of potential primary production map (for scale see Fig. 5).
The map of RPPP was compared with the National Land Degradation Survey (Figure 8). The concentration of high erosion proportions in the Save catchment, around Mutoko, along the Shashe, and around Gokwe all are found in both maps. The National Land Degradation survey was based on the presence of erosion features on aerial photographs, while the RPPP map depends on the reduced productivity of the vegetation compared with its potential for each potential production class. While the two maps indicate related, but different symptoms of the degradation, the degree of agreement increases confidence in the diagnosis of low sustainability in areas with low RPPP.

5.3.3 Indications of advancing degradation in Zimbabwe

The Degradation Survey was at 0.5° resolution and it shows the proportion of area affected by erosion in six classes, unlike the 1km² and percentage values in the RPPP map. To enable quantitative comparison of the two metrics, the RPPP was aggregated to 0.5° resolution and the mean RPPP ±1 standard deviation associated with each class was calculated. Grid cells having RFPP values outside the range of the mean±1 std dev for each Degradation Survey class (Figure 8b) were examined. Those 0.5° grid cells that were in the unaffected or least affected Degradation Survey classes (1 & 2),

---

**Fig. 8. Comparison of degradation mapped using the realized percentage of potential primary production and the National Land Degradation Survey, Zimbabwe (Whitlow, 1988).**

b) Bar charts showing the realized percentages of potential primary production for each of the six erosion classes in the National Degradation map. Mean ± 1 and 2 standard deviations and outliers shown.
c) Differences between the map of realized percentage of potential primary production and National Land Degradation map. Increase means the realized percentage of potential primary production indicated degradation increased, decrease indicates recovery.
but which had an RFPP < (mean - 1 std dev) were identified. Similarly the two most eroded Degradation Survey classes (5 & 6) were identified by selecting cells that had an RFPP > (mean + 1 std dev). The first two difference classes indicate degradation of cells designated not or least eroded in the Degradation Survey, and the second two classes recovery of the most or next to most eroded classes (Figure 8c).

Beyond the differences between the National Land Degradation Survey and the RPPP maps that are simply a result of the different surface properties that they represent, there are clearly areas that were very little affected by erosion in 1980, but that had very low RPPP values (i.e., were degraded) in 2000 (Figure 8c). These include Matibi and Sangwe communal areas in Chiredzi District, Masvingo; Mtetengwe, southern Diti, Dibilishaba, Machuchta, Dendele and Masera communal areas in Matabeleland South; Centenary and Mount Darwin Districts in Mashonaland Central; Siabuwa and Manjolo communal areas in Binga District, Matabeleland North. Thus, either these areas have low RPPP with little erosion, or there has been a serious extension of degradation in the 20 years between the two surveys. Field reports and comparisons with Landsat data indicate there has been an advance of degradation since the 1980s.

6 Discussion

The Local NPP Scaling (LNS) remote sensing mapping technique focuses attention on the areas most likely to be degraded. Reduced productivity highlights potentially degraded areas, which can then be examined further to confirm the diagnosis. Further confirmation is needed since other factors may reduce or increase productivity (e.g., deliberate removal of vegetation, irrigation and water sources other than local rainfall).

A key question regarding the LNS technique is whether it can identify areas where existing maximum productivity is below its productive potential, and thereby highlight areas of desertification. If this capability can be established, the accumulating archives of global remote sensing data may enable the extent and intensity of desertification to be mapped on a grid with a resolution as fine as 1km$^2$. The comparison of the realized percentage of potential production metric with Landsat and the National Land Degradation Survey in Zimbabwe presented above supports this hypothesis. Another question is whether the use of the highest observed productivity, in the case of Zimbabwe over a 4yr period, can estimate the potential productivity. The evidence in support of this second proposition is indirect at this time, based on the observation that the location and degree of estimated degradation in Zimbabwe agreed with that identified by independent sources. A direct test will require comparison with independent measurements of potential production.

Because of the applied nature of the definition of desertification that is adopted here, the appropriate measure of potential productivity is not that of pristine (sometimes called potential) land cover, as would occur in the absence of human management. Rather, cleared land used for pasture or cropping should be considered relative to its potential productivity of goods and services, not the productivity of the land before conversion. Agricultural planners have a long tradition of land evaluation by capability classes (McRae and Burnham, 1981; Food and Agriculture Organization of the United Nations, 1997) and it is the productivity of these classes, in the absence of degradation, which is most relevant.
The LNS method presented here essentially uses a “natural experiment” approach to estimate the potential production of large areas, since conventional ecosystem models lack the forcing data needed to be applied at an appropriate spatial and temporal scale. The existence and ability to detect strata with uniform productive potential is fundamental to the method. Improved stratification into capability classes might be achieved using maps of soil, slope, climate, land cover and cultural features, such as land tenure at appropriate scales. Monitoring degradation and desertification requires the longest possible time series of data. Currently the AVHRR sensor on the NOAA series of meteorological satellites provides a data record starting in 1981 and the MODIS sensor on Terra and Aqua, starting in 2000. Plans are developing to replace the AVHRR with the VIIRS instrument on NPOES, a new series of operational satellites (NOAA 2003). For applications to longer-term ecological processes such as desertification, an instrument record is needed that minimizes the effects of instrument and other changes in the observations to allow the greatest possible sensitivity to changes in the land surface.

Degradation occurs when the productive potential of land is reduced by changed geophysical factors such as drought, or by excessive human utilization, or by both acting together, as may happen, for example, when land is not destocked in periods of drought. There seems to be no point in recognizing this phenomenon with a new term such as “desertification”, if the reduction in productivity caused by drought or over utilization reverses itself fairly quickly when the intensity of land use is reduced or adequate rain returns. This consideration is one reason why the term desertification has fallen into disrepute. It is proposed, however, that there may be circumstances when a distinctive term is appropriate. Four such circumstances have been proposed (Prince 2002) that can be explored using the case study presented here.

First, “desertification” should be reserved for ecosystems that are in stable, less-productive states. The multiple states may have their origin in the internal processes of the vegetation, the landscape, the rainfall, or some combination of these or other processes. The key difference between desertification and reduced productivity caused by reduced rainfall or over-utilization, would be the presence of lags, or quasi-stable states that do not quickly revert to the former state when the proximate causes are removed. An interesting possibility is that the socio-economic system that determines the type and intensity of utilization, may be the source not only of the degradation, but also of the inertia and slow reaction. In this case, improved biogeophysical conditions may not alleviate the desertified condition of the land. Whether all areas with a low realized percentage of potential primary production in Zimbabwe have suffered a shift to a quasi-stable state or states with reduced productivity requires further study, although some such areas certainly exist. The interannual dynamics of the estimated productivity within each pixel should provide some additional insights, as a preliminary to field investigations, in areas highlighted by this analysis.

Second, it is proposed that desertification is a phenomenon that occurs over large space and time scales (Prince 2002). This recognition of scale allows the application of hierarchy theory (O’Neill, DeAngelis et al., 1986). The significance of scale is that it enables state variables and explanatory processes to be differentiated. The exact bounds are less important; a space scale of $\geq 1^\circ \times 1^\circ$ and a time scale of 20-50 years has been proposed for desertification (Prince 2002), so a constraint, such as distance from home to fields, is not confused with the areal extent of interannual variability in rainfall, climate oscillations such as El Niño Southern Oscillations, or
global warming. The case of Zimbabwe is a clear illustration of the fact that the lack of sustainability caused by degradation of land is not only a field scale problem, such as local soil erosion, but it is also a national scale phenomenon caused by different histories of land tenure and governmental policies. The degree of agreement between the 1980 (approximate date) National Land Degradation Survey and the 2000 (approximate date) RPPP map suggests long-term degradation; interannual factors are clearly not responsible for a condition that has persisted for >20 years. The time scale that should be used to calculate the RPPP remains to be determined. It is likely that different time scales should be used for cropped land and forest reserves. Time series analysis of annual productivity may reveal this and other differences between cover types.

Third, human land use is central to the definition of desertification. Desertification focuses on the potential of land to supply goods and services desired by the land manager. In particular, it is the difference between the potential of the land, for whatever purpose it is used in its current state, compared with that in its undesertified state. Furthermore, desertification most usefully applies to land under its existing use, not with respect to a presumed pristine condition before extensive human alteration of the land occurred. The case study of Zimbabwe provides examples of two distinct types of land use that have led to quite different patterns of degradation (Table 2) (Gore, Katerere et al., 1992). The relative contributions of the principal physical factors, drought and land capability, compared with the two principal human land use systems, communal and commercial, provide an unusually clear case for further study.

Fourth, desertification manifests itself in terms of reduced productivity (Reynolds, 2001). In Zimbabwe, there is close agreement of the Land Degradation Survey, based on visual air photo identification of erosion features, with the map of RPPP. It seems probable that reduced primary production can provide an earlier and more sensitive indication of desertification than erosion features. One aspect of productivity that needs further study is the impact of replacement of palatable with unpalatable species in overgrazed rangeland; bush encroachment (Walker, Ludwig et al., 1981) is another. Productivity takes no account of species composition and no reliable method of species identification by remote sensing exists (Hanan, Prévost et al., 1991; Fabricius, Palmer et al., 2002). Whether productivity can detect the same changes that are often defined in terms of changes in species composition remains to be investigated.

The foregoing propositions are all amenable to investigation since they can be stated as questions. Do multiple stable states arise in desertified land? Are humans always involved in the creation of desertified land through over-utilization relative to its productive potential? Are the causes of desertification more clearly understood when particular space and time scales are adopted? Is reduced productivity a key component of desertification? If accepted, each criterion determines the nature of appropriate measurement and monitoring methods for desertification.

Although widespread desertification of semi-arid lands is generally treated as an established fact, this view has been challenged in a number of studies (Watts 1987; Forse 1989; Tucker, Dregne et al., 1991; UNCED 1992; Prince, Brown de Colstoun et al., 1998) and the issue is in urgent need of reconsideration with more appropriate data (Helldén 1991). The actual extent and severity of regional and subcontinental-scale desertification is an important issue that affects policy with respect to economic development strategies and aid programs, and is significant in the context of global
climate change (Rasmussen, Folving et al., 1987) and the global carbon cycle (Schlesinger, Reynolds et al., 1990).

Acknowledgements. This work was supported by NASA grant NAG5 9329 to the author. Ms. Inbal Reshef assisted with the data analyses.

7 References

FEWS. 0.1 degree, ten-day, accumulated precipitation estimates for the portion of the African continent south of 20 degrees North for the period July 1998 to June 1999, Arlington, VA, Created for United States Agency for International Development (USAID) by the Famine Early Warning System (FEWS), 1999.
FEWS/ARD. Zimbabwe Fourth Level Administrative Boundaries (Map). 1:1,000,000, Sioux Falls, SD, USA, Famine_Early_Warning_System-(FEWS), Associates_in_Rural_Development_(ARD), USGS EDC/International Program, 1996.


Watch, H. R. Fast Track Land Reform in Zimbabwe, Human Rights Watch, 2002, 44.


Whitlow, R. Land degradation in Zimbabwe. WRI-122, Harare, Department of Natural Resources, Government of Zimbabwe/Department of Geography, University of Zimbabwe, 1988, 62+Appendices.
1 Introduction

While the environmental impacts of tropical deforestation have received considerable attention, reductions in biomass are in stark contrast to significant increases in woody plant abundance in many grasslands worldwide. Though not well quantified on a global scale, this vegetation change has been widely reported in tropical, temperate and high-latitude rangelands worldwide (Archer 1994; Archer et al., 2001). Land-cover change of this type and magnitude is likely to affect key ecosystem processes in grasslands, and may significantly alter carbon cycling and feedbacks to climate change. Moreover, the proliferation of woody vegetation at the expense of grasses threatens to render substantial portions of these areas incapable of supporting pastoral, subsistence, or commercial livestock grazing, thus adversely affecting 20% of the world’s population inhabiting these lands (Turner et al., 1990; Campbell and Stafford Smith, 2000). While interannual climate variability, atmospheric CO$_2$ enrichment, and nitrogen deposition are also likely contributing factors (Archer et al., 1995; Kochy and Wilson, 2001), land use practices associated with livestock grazing and reductions in fire frequency have been implicated as proximate causes for this widespread land cover change (Archer 1995; Caspersen et al., 2000; Van Auken 2000).

Despite the long-standing recognition of woody plant encroachment as a worldwide natural resources management problem (see Archer 1994), little is known regarding the rates and dynamics of the phenomenon or its ecological consequences. Grassland/savanna systems account for 30 to 35% of global terrestrial NPP (Field et al., 1998). Hence, when woody species increase in abundance and transform grasslands into savannas and savannas into woodlands, the potential to substantially alter C sequestration and dynamics at local, regional, and global scales is great. These changes in vegetation structure significantly affect ecosystem carbon storage because woody plants produce lignin-rich structural tissues that decompose slowly, and they are more deeply rooted than the grass species they displace (Jackson et al., 1996; 2000; Boutton et al., 1999). The result is that organic matter inputs from woody species are more
resistant to decomposition and tend to accumulate in the soil. A significant proportion of this organic matter is distributed deep in the soil profile (Jobbagy and Jackson, 2000) where decomposition and microbial activity are slow (Sombroek et al., 1993). Consequently, grassland→savanna→woodland cover changes have strong potential to increase ecosystem carbon storage and contribute to a global carbon sink (Ciais et al., 1995). Indeed, dryland ecosystems with mixtures of woody and herbaceous vegetation appear to have a higher biodiversity (Solbrig 1996), greater productivity (Long et al., 1989; Scholes and Hall, 1996), and a larger impact on the global carbon cycle (Hall et al., 1995; Scholes and Hall, 1996; Follett et al., 2001) than previously realized. In the most recent USA carbon budget assessments, “thickening” of woody vegetation in dryland and montane forest ecosystems has emerged as a significant but highly uncertain modern sink (Schimel et al., 2000; Pacala et al., 2001; Houghton v 2003a,b). Furthermore, transitions between grass and woody plant-dominated ecosystems may affect regional precipitation patterns (Hoffmann and Jackson, 2000) and concentrations of tropospheric non-methane hydrocarbons (Klinger et al., 1998; Guenther et al., 1999).

The ability to predict changes in landscapes characterized by mixtures of herbaceous and woody plants has emerged as one of the top priorities for global change research (Walker 1996; IPCC 1996a, 1996b; Daly et al., 2000; Jackson 2000; Jobbagy and Jackson, 2000). Quantification of the effects of woody encroachment on C and N sequestration and dynamics is critical to assessing impacts of land-cover/land-use change in grassland ecosystems. However, quantification of such impacts is challenging because woody encroachment rates are highly non-linear and accentuated by episodic climatic events, occur relatively slowly (decadal time scales), across large areas, and in a heterogeneous manner dictated by topography, soils and land use (Archer 1994). To further complicate matters, increases in woody cover in some areas during some time periods may be off-set by episodic wildfire, and climate- (Allen and Breshears, 1998; Fensham and Holman, 1999) or pathogen-induced (McArthur et al., 1990; Ewing and Dobrowolsky, 1992) die-off. These ‘natural’ reductions in woody plant cover are augmented by anthropogenic brush clearing using various combinations of prescribed burning, chemical or mechanical methods, each of which differ in efficacy and treatment longevity (Scifres 1980; Bovey 2001). The extent to which brush management is employed by landowners and land management agencies depends on energy costs, local, state and federal policies, and availability of subsidies. As a result of these combined natural and anthropogenic forces, regional landscapes are dynamic and complex mosaics of woody plant cover classes whose trends through time are difficult to track with traditional technologies and approaches.

Remote sensing can be used to correctly classify vegetation structure and land use and their changes over time. Once these properties are known, we are challenged with interpreting their functional significance with respect to biophysical and biogeochemical feedbacks regulating the climate-atmosphere system. The relative lack of quantitative information regarding the areal extent of these vegetation transformations or their influence on biogeochemical cycles and land surface-atmosphere linkages prompted NASA to fund two projects in the United States that are focused on woody plant abundance in grasslands; one in the humid grasslands of the Great Plains and one in the arid to semi-arid rangelands of the Southwest. The emphasis of both projects was to develop landscape and regional-scale assessment strategies based in detailed, mechanistic field studies. Remote sensing and modeling are being
linked to address the broad-scale properties of these transitional systems, the dynamics of carbon accumulation and loss relative to topo-edaphic and climate gradients, and the strength and influence of the land management legacy on structural and biogeochemical dynamics. These case studies were established to allow us to explore commonalities and differences across grassland systems that suggest key processes or constraints driving the woody encroachment phenomenon, and that will lead us on the path to robust generalizations.

2 Case Studies: the Great Plains and the Southwestern Grasslands of the USA

The relative abundance of grassland versus woodland is governed by interactions between climate (primarily amount and seasonality of rainfall), soils (primarily texture and depth) and disturbances (such as fire, grazing and browsing). The Great Plains and the Southwest regions of the United States consist of grasslands bioclimatically bordered by forest and shrubland, respectively. Slight changes in climate, natural disturbance regimes and/or land use can therefore induce shifts in vegetation structure from grass to woodland (Scholes and Archer, 1997).

Movement of a forest ecotone or expansion of shrubland into neighboring grassland results in dramatic shifts in vegetation structure, from an herbaceous low-stature system to a more vertical, woody state. These features are common to all encroachment scenarios. However, the consequences of encroachment will vary with climate regime, dominant species, and general soil and topographic properties. Vegetation attributes such as plant stature (subshrub vs. shrub vs. tree), leaf longevity (evergreen vs. deciduous) and nitrogen-fixation potential will control functional dynamics, including seasonality of production, nutrient cycling (quality and quantity of litter), and biodiversity. Land management activities which typically affect (directly and indirectly) the extent of grazing and browsing by livestock and wildlife and which seek to control woody plant abundance via prescribed fire or clearing with mechanical techniques or herbicides will disrupt process dynamics and leave landscapes in various states of transition. As a consequence, the short- and long-term impacts of structural shifts from grassland to woodland will vary widely and will be heavily influenced by local management practices.

Despite these system-specific variations, the challenges associated with the remote sensing of woody encroachment fall within a common domain. Influences of canopy geometry, seasonality and background reflectance must be taken into consideration in all scenarios. The relative degree to which these factors confound the detection of changes in woody plant abundance will vary by region, although some generalities will likely fall out along a moisture gradient. For example, in the transition from humid to arid environments, canopy height and woody plant densities will generally decline. As woody vegetation becomes smaller and sparser, the bright background of litter and arid soils will become increasingly evident in satellite imagery. Evergreen and deciduous woody plants are common to both mesic and xeric regions, thus leaf longevity will be species rather than region specific.

Section 3.1 synthesizes results from a series of case studies of ecosystem impacts associated with woody plant proliferation in grasslands in the Great Plains and Southwestern USA. These case studies represent endpoints of a climatic range from mesic/temperate to xeric/subtropical, a woody plant stature gradient ranging from shrub
to arboreal to arborescent and growth-form contrasts (deciduous vs. evergreen, N-fixing vs. non-N-fixing). Section 3.2 then reviews the approaches, pit-falls and challenges in using linked remote sensing-modeling approaches to assess the functional consequences of remotely sensed changes in the structure of ecosystems experiencing shifts in woody versus herbaceous land cover.

3 Assessing the Change and Impacts

3.1 ECOSYSTEM AND BIOGEOCHEMICAL IMPACTS

Extensive databases on productivity, decomposition and nutrient cycling processes exist for grassland, shrubland and woodland ecosystems. However, we cannot necessarily take what we know of patterns and processes in these systems and apply them to systems undergoing shifts from herbaceous to woody plant domination. Furthermore, although many concepts and principles developed for grassland, shrubland and forest systems are potentially relevant, the novel, complex, non-linear behavior of communities undergoing lifeform transformations cannot be accounted for by simply studying or modeling woody and herbaceous components independently (House et al., 2003). The appropriate representation of mixed woody-herbaceous systems is fundamental to the performance of global vegetation models (e.g., Neilson 1995; Daly et al., 2000). Models explicitly incorporating woody-herbaceous interactions and dynamics vary widely with respect to their approach, their complexity and their data requirements. They span a continuum of detail, from highly validated empirical formulations to mechanistic, spatially-explicit treatment of individual plants and vary with respect to (a) the extent to which they incorporate plant physiological and population processes; (b) their fundamental assumptions of how and to what extent woody and herbaceous plants access, utilize, and redistribute resources; (c) their spatial and temporal resolution; (d) the extent to which they incorporate effects of climate, soils and disturbance; and (e) their treatment of competition or facilitation interactions.

As discussed in subsequent sections, linking remote sensing of changes in woody plant cover/biomass/leaf area with ecosystem process models is one approach for making large scale assessments and predictions of changes in ecosystem function resulting from changes in the relative abundance of woody plants in dryland systems.

Table 1 summarizes some of the changes known to occur when woody plants invade and establish in grasslands. What is remarkable from these studies is the speed at which some changes have occurred. Changes in soil properties for example have traditionally been viewed as occurring on the scale of centuries. However, as several of the studies in Table 1 indicate, significant changes in soil organic carbon (SOC) and nitrogen pools can occur at decadal time scales subsequent to the establishment of woody plants in grasslands. Indeed, carbon inputs from woody plants appear to be dominating the SOC pool in upper horizons within 50 years of their establishment (See ‘Surficial Soil C from Woody Vegetation’ in Table 1). SOC mass reflects the balance between organic matter inputs from plants and losses from the decay of organic matter. In the context of woody plant proliferation, SOC could increase if woody plants were more productive than herbaceous plants, and/or if woody-plant tissues decayed more
slowly than herbaceous plant inputs. Precipitation and temperature mediate this trade-off by exerting control over both plant growth (inputs) and decomposition (outputs).

Table 1. Changes in ecosystem properties accompanying woody plant encroachment into grasslands (−−, −, 0, +, ++ represent substantial decrease, slight decrease, no change, slight increase, and major increase, respectively; “?” denotes expected but unsubstantiated changes). Numerical superscripts point to studies reporting these changes; letter superscripts refer to explanations (see footnotes). For additional information on sites, visit the following URLs: Konza: http://climate.konza.ksu.edu/HomePage.html Vernon: http://juniper.tamu.edu/IRM/brush/P01JAprojecthome.htm La Copita: http://www.geocities.com/lacopita_research_area/ Jornada: http://usda-ars.nmsu.edu/ Sevellita: http://sevilleta.unm.edu/; Santa Rita: http://ag.arizona.edu/SRER/

<table>
<thead>
<tr>
<th>Metric</th>
<th>Konza</th>
<th>Vernon</th>
<th>La Copita</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lat/Long</td>
<td>39.1°N; 96.1°W</td>
<td>34.5° N; 99.2° W</td>
<td>27.4° N; 98.1° W</td>
</tr>
<tr>
<td>City</td>
<td>Manhattan</td>
<td>Vernon</td>
<td>Alice</td>
</tr>
<tr>
<td>State</td>
<td>Kansas</td>
<td>Texas</td>
<td>Texas</td>
</tr>
<tr>
<td>Annual PPT (mm)</td>
<td>835 mm</td>
<td>655 mm</td>
<td>680 mm</td>
</tr>
<tr>
<td>Annual Mean Temp (˚C)</td>
<td>13 ˚C</td>
<td>17 ˚C</td>
<td>22 ˚C</td>
</tr>
<tr>
<td>Characteristics of Dominant Woody Plants</td>
<td>Tree</td>
<td>Tree</td>
<td>Tree</td>
</tr>
<tr>
<td>Stature</td>
<td>Evergreen</td>
<td>Arborescent</td>
<td>Arborescent</td>
</tr>
<tr>
<td>Genera</td>
<td>Juniperus</td>
<td>Deciduous</td>
<td>Deciduous</td>
</tr>
<tr>
<td>Potential N₂-fixation?</td>
<td>No</td>
<td>Yes</td>
<td>Yes</td>
</tr>
<tr>
<td>Fractional Cover Δ (%/y)</td>
<td>+2.3%/yr</td>
<td>+0.2-2.2%/yr</td>
<td>+0.7%/y</td>
</tr>
<tr>
<td>Soil Temp Δ</td>
<td>--</td>
<td>--</td>
<td>--</td>
</tr>
<tr>
<td>Surfacial' Soil Moisture Δ</td>
<td>-? 7</td>
<td>0, + 10</td>
<td>+ 7 11</td>
</tr>
<tr>
<td>ANPP Δ</td>
<td>++ 12</td>
<td>++ ?</td>
<td>++ 13</td>
</tr>
<tr>
<td>Plant C Pool Δ Aboveground</td>
<td>++ 12</td>
<td>++ 13 2</td>
<td>++ 13</td>
</tr>
<tr>
<td>Plant C Pool Δ Belowground</td>
<td>0? 7</td>
<td>+ 11 18</td>
<td>+ 11 18 38</td>
</tr>
<tr>
<td>Soil Organic Carbon</td>
<td>0 19</td>
<td>- + c 15 20</td>
<td>++ 9 11 18 38</td>
</tr>
<tr>
<td>Surfacial' Soil C from Woody Vegetation (%)</td>
<td>50% 19</td>
<td>45-88% 24</td>
<td></td>
</tr>
<tr>
<td>Soil Respiration Δ</td>
<td>-- 7</td>
<td>++ 9</td>
<td>++ 9</td>
</tr>
<tr>
<td>Nmin Δ</td>
<td>+? 27</td>
<td>++ 11</td>
<td>++ 11</td>
</tr>
<tr>
<td>NO/N₂ Flux Δ</td>
<td>+ 28</td>
<td>++ 29</td>
<td>++ 29</td>
</tr>
<tr>
<td>NMHC Flux Δ</td>
<td>++ 31</td>
<td>++ 9</td>
<td>++ 9</td>
</tr>
<tr>
<td>Microbial Biomass Δ</td>
<td>0 7</td>
<td>++ 9</td>
<td>++ 9</td>
</tr>
<tr>
<td>Potentially Mineralizable Soil C</td>
<td>0 7</td>
<td>++ 9</td>
<td>++ 9</td>
</tr>
<tr>
<td>Net C exchange (source, sink)</td>
<td>Sink (?) 33</td>
<td>Sink 34</td>
<td>Sink 13</td>
</tr>
<tr>
<td>Litter Decomposition</td>
<td>-- 35</td>
<td>-- 18</td>
<td>-- 18</td>
</tr>
<tr>
<td>Maximum depth of nematodes (m)</td>
<td>0 20</td>
<td>++ 9</td>
<td>++ 9</td>
</tr>
<tr>
<td>Plant Species Diversity (richness)</td>
<td>-- 36</td>
<td>0, + (?) 16</td>
<td>0, + (?) 16</td>
</tr>
</tbody>
</table>
The broad range of responses in Table 1 likely results from several factors. First, changes in SOC may be species dependent based on plant productivity, allocation patterns, and/or tissue chemistry. For example, at the Jornada site (MAP = 230 mm) SOC increases ~230% under tarbush, but decreases under creosote bush (-9%) and...
mesquite (-15%) (Schlesinger and Pilmanis, 1998). In Texas (Vernon and La Copita) (MAP = 660-715 mm), SOC increases following woody plant proliferation in former grasslands range from 9% in temperate mesquite stands (Hughes et al., 2000) to 27-103% in more diverse subtropical woodlands (Boutton et al., 1998). Woody plant effects on microclimate which affect decomposition rates (notably soil moisture and temperature) also vary among growth forms in that evergreen and deciduous canopies differ in their magnitude and seasonality of rainfall and radiant energy interception, potentially affecting decomposition processes and hence C and N pools and fluxes.

![Diagram of soil respiration and litterfall]

Figure 1. Summary of measured carbon stocks (kg/ha) and fluxes (kg/ha/yr) in grassland and closed canopy Juniper forest. Note large increase in C stocks on forested sites for all strata. Soil respiration in forest is reduced by ~30% compared to grassland. (Smith and Johnson, 2003a)

Differences in woody plant effects on soil properties listed in Table 1 might also reflect differences in the ways microbial communities respond to changes in vegetation structure. For example, shifts from bacterial to fungal populations may accompany shifts from herbaceous to woody domination (Purohit et al., 2002), thus enabling decomposers to more effectively deal with lower litter quality (i.e. increasing C:N), and hence maintain or increase soil respiration and mineralization. Changes in root biomass distribution accompanying shifts from grass to woody plant domination may also change the nature and depth of microbial activity, but available information available is scanty and conflicting. Jackson et al. (2002) inventoried nematodes dependent on plant roots as indicators of changes in microbial activity accompanying shifts from grass to woody plant dominance. They found substantial decreases in maximum depth on some sites (Jornada), substantial increases in maximum depth on some sites (Sevilleta) and no changes on another site (Vernon).

It is interesting to note that plant and soil C and N stocks increase at some sites despite significant increases in soil respiration, non-methane hydrocarbon emissions, N-mineralization and NO emissions (e.g. La Copita, Santa Rita). In contrast, Juniperus encroachment in Konza appears to have caused little change in the SOC pool, despite suppression of soil respiration and high inputs of low litter quality by this evergreen arborescent (Figure 1) (Smith and Johnson, 2003a). Explanations for this behavior are elusive. At the Vernon site, some studies have shown significant
declines in SOC with woody plant encroachment (Jackson et al., 2002) while others have shown significant increases (Hughes et al., 2000). Reasons for this discrepancy may be indicative of the importance of local differences in soil types and land management histories (e.g. Teague et al., 1999; Asner et al., 2003). Indeed, topo-edaphic features do exert substantial control over the direction and rate of change in plant and soil nutrient pools and fluxes. For example, SOC increases in subtropical woodland communities developing on former grasslands vary from 27-37% on upland sandy loam soils to 103% on lowland clay loam soils (Boutton et al., 1998).

The potential for ecosystem C-sequestration associated with the conversion of grass- to woody plant domination will also reflect that balance between biotic processes promoting carbon accumulation (plant modification of soils and microclimate) and geophysical processes promoting nutrient losses (wind/water erosion). The magnitude of geophysical-induced losses and extent to which woody plants can compensate for these likely varies with soils and climate. For example, disturbances such as grazing, which promote woody plant encroachment, may also accelerate the loss of SOC via increased oxidation and erosion. At the Jornada site, shifts from grass to shrub domination have caused major changes in soil nutrient distributions (nutrient pools in shrub-affected soils >> nutrient pools of non-shrub soils), but no net change in total carbon stocks at the landscape scale, as C gains associated with woody plant proliferation are relatively small and have been offset by losses from inter-shrub zones (Connin et al., 1997; Schlesinger and Pilmanis, 1998). In contrast, at the La Copita site, losses of SOC associated with livestock grazing in the late 1800s-early 1900s appear to have been fully compensated for by invading woody plants by the 1950s; and by the 1990s, landscapes had ca. 30% more carbon than would occur had the pristine grasslands, present at the time of settlement, been maintained (Hibbard et al., 2003). These contrasting scenarios point to the need to account for both loss and gain vectors and to the potential dangers of extrapolating from plant or patch scale measurements to ecosystem/landscape scales.

Studies documenting effects of woody plant encroachment on ecosystem processes are accumulating. However, an overlooked aspect of the woody plant encroachment phenomenon is the fact that land managers have been and will continue to implement management practices to reduce woody plant cover (see Section 4.1). Currently we know little of the extent of such clearing practices, rates of woody community recovery following treatments, or how the treatments affect soil nutrient pools and fluxes.

The challenge for the remote sensing community is to provide tools for tracking structural and biophysical changes accompanying shifts in woody versus herbaceous plant abundance. The challenge for ecosystem modelers is to develop approaches for representing and predicting, in a spatially explicit fashion over large areas, the ecosystem specific changes (Table 1) that occur when land cover transitions from grass to woody plant domination. The linkage of remote sensing and ecosystem process models appears to be a viable strategy for tracking the functional consequences of changes in the relative abundance of herbaceous and woody vegetation in transitional grasslands. Approaches for doing this are discussed in the next section.
3.2 LANDCOVER CHANGE

3.2.1 Remote Sensing of Grassland-Woodland Transitions

Conversion of grasslands to shrub- or woodlands creates heterogeneous landscapes that challenge most remote sensing techniques seeking to quantify land-cover types and land-cover change. Pure pixels of one structural type are rare, and mixtures of soil, litter, wood, foliage, and canopy geometries are the norm. Our discussion of remote sensing science centers on the remote sensing of drylands due to our experience and progress, and the unique challenges these systems present for vegetation monitoring. Arid and semi-arid regions contain some of the most complex spatial mixtures of vegetation, soil and rock material on Earth. These drylands extend over about 45% of the global land surface and are subject to a vast array of land uses and climate variations. Field-based ecological, hydrological and geological studies are thus very tenuous without the bird’s eye view afforded by airborne and space-based remote sensing. Remote sensing techniques developed for woody encroachment assessment in drylands can be applied to more mesic environments such as the Great Plains in which heterogeneity in cover types and land uses is also prevalent. In general, the same principles will apply to these areas as the drylands, with noted exceptions resulting from changes in background (increased ground cover) and, in some cases, seasonality.

A wide variety of studies have employed optical, passive microwave and active (e.g., LIDAR) observations in the pursuit of spatial and temporal information on ecosystem dynamics. Optical radiance or reflectance approaches have been the most successful. Here we summarize the challenges and successes in developing airborne and space-based optical remote sensing methods for quantifying vegetation cover in heterogeneous landscapes. We emphasize studies, including our own, that address the issue of quantifying woody and herbaceous plant canopy properties, as this information is central to any analysis of woody encroachment and cover change.

Vegetation cover is arguably the most important remote sensing measurement needed to extend a field-level understanding of ecological, hydrological, and biogeochemical processes to broader spatial and temporal scales. It is critical for regional-scale monitoring of land management practices (e.g., Pickup and Chewings, 1994; Pickup et al., 1994); and it serves as an important indicator of ecological and biogeochemical processes (Table 1; Schlesinger et al., 1990, 1996; Archer et al., 2001). Furthermore, cover information is needed to constrain ecosystem and land-surface biophysical models to actual abundance and distribution of cover types (e.g., Running et al., 1994; Sellers et al., 1997; Neilson 1995; Daly et al., 2000).

Remote sensing of vegetation “condition” is another important component of the effort to monitor changes in land cover and use. Here, we define vegetation condition as the vigor, photosynthetic capacity, or stress of a given vegetation canopy. Variations in vegetation leaf area index (LAI), fraction of photosynthetically active radiation absorbed (fAPAR), and water content indicate plant condition, and can be estimated from remotely sensed optical data (e.g., Asrar et al., 1986, 1992; Ustin et al., 1998; Qi et al., 2000). These characteristics can be indicative of both plant function and land use intensity (e.g., Pickup et al., 1994; Asner et al., 1998b). Integrated over time, fAPAR can be used to estimate net primary productivity (Prince 1991; Field et al., 1995), providing an avenue to extend field-level relationships between plant productivity and other ecosystem processes to broader spatial scales. Remotely sensed estimates of standing litter or “dry carbon” cover, content and biomass also indicate
vegetation condition (Wessman et al., 1997; Asner and Lobell, 2000) and provide an
index of fire fuel loads, flammability, hydrological function, and carbon cycling (Asner
et al., 1998b; Roberts et al., 1998).

3.2.2 Spatial Observations
Spatial observations are the oldest and most intuitive type of remote sensing. Today,
there is a significant demand for high spatial resolution data such as from aerial
photography and spaceborne sensors. The newest instruments such as the satellite
IKONOS (http://www.spaceimaging.com) provide monochromatic imagery at <1m
spatial resolution. Schlesinger and Gramenopoulos (1996) used declassified high
spatial resolution (~4m) monochromatic reconnaissance satellite photographs to
estimate changes in woody vegetation cover between 1943 and 1994 along the Sahel-
Sahara Desert ecotone in west Sudan. Their findings showed no change in woody plant
canopy cover following widespread drought in mid- to late twentieth century. Asner
and Heidebrecht (2002) used IKONOS imagery at the Jornada Experimental Range in
New Mexico (USA) to quantify woody vegetation cover. IKONOS-based results
agreed well with both field and low-altitude aerial photography estimates of woody
canopy cover. However, the ~1 m IKONOS data were valuable for quantifying woody
cover only when the canopies were ≥3 m in diameter.

Landscape or image texture is closely related to the issue of spatial resolution.
Texture refers to the local variation of land surface components such as shrubs and bare
soils, both in terms of percentage cover and spatial distribution. In remotely sensed
data, texture statistics describing the local variation of pixel brightness (mean, variance,
and range of values within a specified pixel window) for each location can be analyzed
across a large geographic region to estimate variation in the vegetation and soil cover.
Image texture provides a means to understand land cover heterogeneity and the changes
that occur at a spatial scale commensurate with human activities (Haralick et al., 1973;
Franklin and Peddle, 1990). This approach also provides a means to analyze historical
aerial photographs by minimizing the effects of systematic errors associated with
background brightness variation and vignetting. Hudak and Wessman (1998, 2001)
used textural filtering of digitized aerial photographs and geostatistical analyses to
estimate shrub density and temporal variability in South African savanna landscapes
over a 30-year period.

3.2.3 Spectral Observations
Spectral radiances (or reflectance) observations contain significant information on
vegetation and soil properties of ecosystems. One of the most common means to
remotely assess vegetation characteristics is the Normalized Difference Vegetation
Index (NDVI), a metric used to detect changes in pixel-scale vegetation greenness. The
NDVI is broadly correlated with canopy chlorophyll and water content (Sellers 1985;
Moran et al., 1989; Ustin et al., 1998) and as such can be linked to LAI and fAPAR by
plant canopies (e.g. Asrar et al., 1986, 1992; Myneni and Williams, 1994, and others).

The NDVI has been the most commonly employed spectral index in dryland
environments. For example, cattle pasture condition in drylands has been assessed by
linking the NDVI to field estimated canopy greenness, cover and biomass (Williamson
and Eldridge, 1993). Time series of greenness derived from the NDVI have been
successfully employed in mapping vegetation community and physiological classes in
temperate grasslands and shrublands (Paruelo and Lauenroth, 1995; Tieszen et al.,
Gamon et al. (1993, 1995) found strong correlations between the NDVI and leaf area index, aboveground biomass, canopy nitrogen and chlorophyll content, and green canopy fAPAR in California grasslands. The NDVI has been used to assess vegetation greenness changes associated with large-scale rainfall anomalies in dryland regions (Nicholson et al., 1990; Tucker et al., 1991), including those related to El Niño-La Niña cycles (Myneni et al., 1996). Although the NDVI is sensitive to pixel-level changes in greenness and fAPAR, it is not differentially sensitive to changes in vegetation cover versus condition (Carlson and Ripley, 1997). When an NDVI change occurs, whether or not it was caused by altered vegetation cover or condition of the cover cannot be readily determined. Moreover, the NDVI has had limited success in providing accurate estimates of shrubland cover in arid regions (e.g., Duncan et al., 1993), owing to the variability of background materials such as soils and surface litter (Huete and Jackson, 1988; van Leeuwen and Huete, 1996). We conclude that the NDVI alone is not sufficient for quantifying woody canopy cover in drylands.

Multi-spectral, non-NDVI measurements have been developed to estimate vegetation cover in drylands. Pickup et al. (1994) used a multi-temporal vegetation index derived from visible wavelength channels to successfully estimate semi-arid rangeland vegetation cover. One of the most common methods for woody and herbaceous cover analysis of grasslands involves decomposing image pixels into their constituent surface cover classes. Known as spectral mixture analysis (SMA), this method allows for the estimation of biophysiologically distinct cover types at the sub-pixel level. A wide range of SMA efforts have now been applied in analyses of grasslands using airborne and spaceborne multi-spectral scanners (e.g., Graetz and Gentle, 1982; Smith et al., 1990; Wessman et al., 1997; Asner et al., 1998a; Elmore et al., 2000).

A major assumption in linear mixture modeling is that the spectral variability of the major landscape components is accommodated by the reflectance signatures employed in the models. Some SMA approaches utilize spectral endmembers derived from the image (e.g., Wessman et al., 1997; Elmore et al., 2000), while others employ libraries of endmember spectra (e.g., Smith et al., 1990a,b; Roberts et al., 1998). In heterogeneous landscapes, it is exceedingly difficult to locate image pixels containing 100% cover of each pertinent endmember, which is usually required when using image-derived endmembers in a spectral mixture model. Thus, library spectra have been widely employed with the recognition that libraries cannot easily capture the full range of endmember variability as is found in nature. Bateson and Curtiss (1996) and Bateson et al. (2000) developed a unique SMA model that allows for the exploration of image data in multiple dimensions via principal components analysis. The technique allows the user to select endmember spectra based on the inherent spectral variability of the image data without requiring homogeneous pixels of each endmember.

Independent of the endmember selection technique, Landsat-type instruments tend to provide sufficient spectral information to broadly discriminate between green vegetation and non-photosynthetic materials such as litter and soil (Smith et al., 1990; Asner et al., 1998a). However, they do not typically provide the spectral resolution necessary to delineate species, functional groups, or greenness conditions within the “green vegetation” class using spectral mixture models unless seasonality enables such separations. In a study to estimate *Juniperus virginiana* (Eastern redcedar) canopy abundance in eastern Kansas, Price (unpublished data) unmixed Landsat TM imagery acquired at a time when much of the matrix of tallgrass and deciduous forest was in the dormant stage. The difference between green and non-photosynthetic vegetation helped
to distinguish the redcedar from the background when redcedar coverage exceeded 20% (Figure 2). See CD for color image.

Figure 2. Redcedar cover in the study area estimated from spectral mixture analysis of Landsat 7 ETM+ imagery. While redcedar invasion into the grassland is pervasive, only a very small area (<1%) in the study areas are covered with close redcedar (redcedar fraction > 80%). Most areas have redcedar coverage between 25% - 50%, indicating the relative new encroachment of the woody species. The large area with less than 25% redcedar in this study area strongly implies that redcedar cover is overestimated at low values. Further work is needed to refine the SMA model to reduce the uncertainty at low redcedar coverage estimate. (Price, unpublished data)

Multi-spectral sensors such as Landsat TM and MODIS may not provide sufficient information to spectrally separate soils from non-photosynthetic vegetation (Asner et al., 2000). The performance of linear spectral mixture analysis has been compared to vegetation indices in drylands using multi-spectral satellite data. Elmore et al. (2000) compared the performance of a spectral mixture model against the NDVI in mapping green canopy cover from Landsat data. Although the NDVI was generally correlated with green cover, a marked increase in performance was obtained when utilizing the full multi-spectral data from Landsat with spectral mixture analysis. Similarly, McGuire et al. (2000) demonstrated that SMA was more accurate than the NDVI (and other indices) for quantifying green canopy cover in a California desert.

Limitations to the information contained in multi-spectral imagery led to the evolution of higher spectral resolution imagers in the 1980s and 1990s. Today, imaging spectroscopy employs hyperspectral data to quantify the spatial extent, biochemistry, and geochemical properties of materials. Imaging spectrometers have recently become available for use from Earth orbit; the NASA EO-1 Hyperion instrument is the first full spectral range imaging spectrometer to measure Earth’s spectral properties from space. The additional information provided by hyperspectral imagers over that of multi-spectral sensors has advanced many analyses of drylands. For example, using spectral unmixing techniques, Wessman et al. (1997) related subtle differences in hyperspectral reflectance endmembers to biophysical conditions related to rangeland management in a Kansas grassland. In particular, high spectral resolution allowed separation of litter from soil based on plant lignin-cellulose absorption features.
Several other efforts have combined hyperspectral reflectance data with spectral mixture models to estimate sub-pixel cover of vegetation in drylands. Roberts et al. (1998) used a multiple endmember spectral mixture model to map major plant functional groups and species in a California chaparral ecosystem. Asner and Lobell (2000) used shortwave-IR (2000-2500 nm) hyperspectral data from AVIRIS to accurately estimate green vegetation, non-photosynthetic vegetation and bare soil extent in arid shrublands and grasslands of the Chihuahuan Desert, New Mexico, USA. In addition, Asner et al. (1998b) used imaging spectrometer data with spectral mixture analysis and radiative transfer inverse modeling to estimate both the horizontal extent and vertical density of live and senescent vegetation and fire fuel load in subtropical savanna ecosystems in southern Texas.

3.2.4 Angular Observations
Angular reflectance properties of vegetation and soils have been assessed in many different regions, including grasslands (Deering et al., 1990, 1992). All studies show canopies to be directionally non-uniform or anisotropic reflectors of solar energy. Reflectance anisotropy means that the reflectance behavior of vegetation can vary solely by differences in the angle from which the surface is observed or illuminated. Another term commonly used to describe the angular reflectance behavior of a surface is the “bidirectional reflectance distribution function” or BRDF. The angular reflectance behavior of any land surface is determined predominantly by the spatial distribution and fractional amount of shadow apparent to the observer or remote sensor. Therefore, it is another signature domain providing additional information on the biophysical properties of ecosystems.

The shape of the BRDF of land surface results from the orientation of the photon scattering elements (e.g., canopy foliage, shrub crowns) in 3-dimensional space (Ross 1981). The BRDF of grasslands and woodlands is therefore dependent upon the orientation of foliage, litter and wood at canopy and landscape scales and by the roughness of the exposed soil surface (Li and Strahler, 1985; Pinty et al., 1998). Leaf, litter and wood inclination and azimuthal orientation play a major role in determining the angular reflectance behavior of individual or horizontally homogeneous canopies (Myneni et al., 1989; Qin 1993). At the landscape level, the spatial distribution and shape of individual canopies or crowns can account for characteristic variation of the angular reflectance (Strahler and Jupp, 1991; Li and Strahler, 1985, 1992). For example, the number and size of gaps between shrubs in arid ecosystems can significantly affect landscape-level shadowing (Franklin and Turner, 1992) and thus shape of the BRDF. The landscape structural properties affecting shadow can dominate the observed variability in angular reflectance signatures.

The NASA FIFE (Sellers et al., 1988), NASA PROVE (Privette et al., 2000), and international HAPEX-Sahel (Goutorbe et al., 1997) experiments highlighted the issue of surface reflectance anisotropy in drylands. Studies showed that up to 80% of the variability in the remote sensing measurements acquired by the NOAA AVHRR and many field sensors was due to surface BRDF effects (e.g. Deering et al., 1992; Privette et al., 1996; Sandmeier et al., 1999). A common goal of these efforts was to step from treating vegetation reflectance anisotropy as noise to treating the angular signatures as useful information. For example, Diner et al. (1999) showed that a set of six viewing angles of surface radiance improved LAI estimates over that which could be acquired using single view angle nadir observations. Privette et al. (1996) and Gao
and Lesht (1997) used similar techniques to improve estimates of LAI in Kansas grasslands.

Sampling the surface BRDF can also be used in other ways to improve the accuracy of more traditional satellite metrics such as the NDVI in dryland regions. Chopping (2000) used a BRDF model to adjust for vegetation-specific reflectance anisotropy effects on red and NIR observations from a ground-based radiometer. By accounting for the view-angle dependence of reflectance among ten unique vegetation types in semi-arid ecosystems of Mongolia, he was able to significantly improve the spectral separability and subsequent classification of vegetation covers in the region. Asner et al. (1998a) used multi-angle AVHRR observations to account for shadow fraction in Landsat imagery, allowing the accurate estimation of woody canopy cover and leaf area index in Texas savannas.

3.2.5 Historical Woody Cover Change Analysis

There are numerous trade-offs between using aerial photography or satellite imagery to track changes in woody plant cover in grassland to woodland transitions. Aerial photos, which may date back many decades, are relatively inexpensive and can provide a deeper historical baseline from which to document change than satellite imagery, which dates back only to the 1970s. In addition, the spatial resolution of aerial photos is often more commensurate with the ground area occupied by the vegetation of interest (e.g., individual trees or shrubs), thus requiring little in the way of image manipulation. In contrast, satellite data require sophisticated calibration efforts, and the greater disparity between satellite spatial resolution and vegetation patch characteristics requires analytical techniques such as spectral mixture analysis. There is also a trade-off in ascertaining large-scale changes in woody plant cover in grasslands: the labor-intensive process of developing mosaics of very high resolution aerial photos versus using lower spatial resolution satellite imagery covering a much larger geographic area but requiring more complicated signal processing and ground validation efforts.

As a compromise, Asner et al. (2003) opted to use a mosaic of high-resolution aerial photos to establish an historical baseline for woody vegetation cover and satellite imagery to quantify contemporary cover. They used this combination to quantify woody cover and aboveground carbon changes for a 63-year period in a north Texas rangeland. Mosaics of high spatial resolution aerial photography were analyzed for woody cover in 1937 using textural filtering and classification techniques. Areal estimates of woody cover in 1999 were then quantified using Landsat 7 data with spectral mixture analysis.

Comparison of the 1937 and 1999 imagery revealed major changes in woody plant cover and aboveground carbon (Figure 3). There were numerous landscapes throughout the region where woody cover increased from < 15% in 1937 to > 40% in 1999. There were also substantial areas where woody plant cover decreased from > 80% in 1937 to < 50% in 1999. The result was a net increase in woody cover and homogenization of woody cover over the 63-year period. Reasons for declines in woody cover on many landscapes in the region are discussed in Section 4.1.
4 Challenges and Caveats

4.1 SCALE-DEPENDENCE OF OBSERVATIONS

Grassland landscapes undergoing woody encroachment are heterogeneous in both space and time. Detection of shrubs and trees within a grass matrix require image resolutions commensurate with the scale of the woody plants or sub-pixel analyses such as spectral mixture analysis. Similarly, frequency of data acquisition, remote and field-based, will have significant influence on analyses and interpretation of cover dynamics and their biogeochemical consequences. The shrub encroachment process under “natural” conditions progresses on a decadal scale. However, management practices introduce a temporal complexity to the landscape as different areas or management units experience different land uses at different times. For example, pastures or portions of pastures with high woody cover may be targeted for ‘brush management’ and those with low woody cover excluded from treatment. Brush may be cleared via mechanical means in some pastures and via herbicides or prescribed fire in others.

Figure 3. Changes in woody vegetation cover in northern Texas between 1937 (left) and 1999 (right). Cover in 2937 was estimated from aerial photography and texture-based classification convolved to 30 m spatial resolution; cover in 1999 is from Landsat 7 ETM+ imagery. White lines denote fence lines separating livestock management units on the Kite Camp portion of the Waggoner Ranch. (Asner et al., 2003).
A conceptual model illustrating the challenges to assessing regional woody plant cover and dynamics in the context of brush management is presented in Figure 4. Line I represents woody stand development that might occur in the absence of disturbance (e.g. elimination of fire due to grazing or active suppression) or management intervention. Line II represents a stand whose development is interrupted by natural (e.g. drought (Archer et al., 1988; Allen and Breshears, 1998), wildfire (Kurz and Apps, 1999), pathogenic (McArthur et al., 1990; Ewing and Dobrowolski, 1992) or anthropogenic (e.g. brush management (Scifres 1980, Bovey 2001)) events that ‘reset’ the carbon accumulation process. The magnitude of these setbacks and rates of recovery vary depending upon the type, intensity and spatial extent of disturbance, soil type, environmental conditions immediately preceding and following the disturbance, and the growth form (evergreen vs. deciduous) and regenerative traits involved. Some stands regenerating from these setbacks might receive follow-up brush management treatments (Line III), but others may not due to financial constraints, availability of subsidies, and many other factors. Thus, remote sensing observations over large areas and limited temporal resolution show net changes (A), whereby increases in woody cover on some landscapes or management units (B and C) are offset by decreases in others (D).

In sum, remote sensing analyses of woodland expansion must be compatible with the spatial scale of the landscape components and the temporal resolution of the dynamics driving cover changes. Frequency of data acquisition must keep pace with disturbance dynamics and/or land use change in order to capture the important transitional stages associated with management and recovery processes. For example, measurements with poor temporal resolution of net changes in woody plant cover across long time periods may insufficiently estimate rates of carbon cycling and consequently the source/sink potential of an area under transformation.
4.2 REMOTE SENSING-MODELING LINKS

New generations of ecosystem process models that incorporate remote sensing products as a basis for spatially explicit calculations at large scales are at various stages of development. Approaches linking dynamic simulations of function and process to remote sensing of structure and pattern hold promise for assessments of the functional consequences of changes in land-use/land-cover at unprecedented spatial and temporal scales. (e.g. Field et al., 1995; Schimel et al., 1997; Wylie et al., 2003). For example, in models such as the Carnegie-Stanford Approach (CASA), calculations of NPP are based on remote sensing-estimates of APAR rather than mechanistic details of NPP (Field et al., 1995). This constrains the calculations to observed heterogeneity and reduces errors resulting from unrealistic assumptions based on optimum or potential conditions. This point is particularly important under conditions of woody plant encroachment, in which fundamental shifts in vegetation form result in profound functional differences and transitional properties that cannot be easily estimated based on a steady-state modeling approach. Even a simple modeling exercise exploring diurnal PAR absorption and carbon uptake in a Texas savanna found that LAI, vegetation structure, and intercanopy shading (all estimated remotely) are important controls on carbon fluxes which may scale to affect regional carbon estimates (Asner et al., 1998a).

In spatially heterogeneous environments, integration of remotely sensed data with ecosystem models enables us to establish a fundamental connection between the spatial structure and the manifestation of functional processes at landscape scales, an association that is difficult to achieve based solely on field measurements under the best of conditions (Wessman and Asner, 1998). Even if we were able to use field-based approaches, the sheer vastness and remoteness of the world’s drylands would make it impossible to make such assessments at the frequency and degree of spatial coverage that would be needed to adequately assess and track land use-land cover changes. Remote sensing not only provides access to the spatial distribution of vegetation structure, but also provides some means to bypass our present-day inability to mechanistically connect principles of allocation to biogeochemistry and ecosystem function (Wessman and Asner, 1998). Through the integration of remote sensing and modeling, we can, to some degree of accuracy, calculate and track NPP and both above- and belowground (e.g. Gill et al., 2002) carbon storage and dynamics under contrasting land use practices at landscape and regional scales.

5 Conclusions

Although shifts from grass to woody plant domination have been widely reported in the world’s grasslands (Archer et al., 2001), there has been no effort to systematically quantify the rate or extent of change nor to evaluate its biogeochemical consequences at large scales. Two recently emerging factors add urgency to this particular land cover change issue: (a) the latest USA carbon budget assessments which implicate “thickening” of woody vegetation in grasslands as a major (Houghton 2003a) or perhaps even the single largest sink term (Schimel et al., 2000); and (b) the possibility of industry or government-sponsored “carbon credit” or “carbon offset” programs. Jackson et al. (2002) stress that current uncertainties around the net change in the
carbon cycle due to woody encroachment are large, as are the uncertainties in regional extrapolations of the biogeochemical consequences. Indeed, the complexity in such broad functional shifts in grassland to woodland transitions coupled to socioeconomic drivers of change are profound and in need of further study.

Our studies of grasslands in the Southwest and Great Plains emphasize the importance of three factors. First, the encroachment phenomenon is of sufficient magnitude and extent that synoptic monitoring via remote sensing of the spatial distribution and temporal dynamics of woody plant abundance is imperative. The ecosystem impacts of grassland to woodland transitions cannot be captured by ground measurements alone. However, and second in our list, studies of the biogeochemical consequences of these transitions must recognize the importance of understanding local and landscape mechanisms in order to achieve accurate and prognostic regional assessments. This requires well-designed field studies, documentation and monitoring of land use practices, and the implementation of ecosystem simulation models to test our knowledge and build scenarios of change trajectories. We emphasize the importance of integrating fieldwork into the analysis and interpretation of remote sensing data and model development to achieve sufficient understanding of these complex landscapes.

A third important factor is the fact that, traditionally, there has been strong policy, subsidy and economic incentives for brush clearing on rangelands. Indeed, brush management is often the greatest single expense in commercial ranching enterprises (Scifres 1980; Scifres and Hamilton, 1993; Bovey 2001). However, with the prospect of carbon credit/offset programs, ‘brush’ may become an income-generating commodity because of its potential to sequester more carbon above- and belowground relative to the grasslands it replaced (e.g., Archer et al., 2001). It is easy to envision scenarios in the near future, whereby land owners/managers may be paid NOT to clear existing woody vegetation. Furthermore, there could be strong economic incentives to engage in land management practices that promote woody plant encroachment and the displacement of grasslands. From a carbon sequestration perspective this may be desirable. However, perverse outcomes with respect to livestock production, wildlife habitat, grassland biodiversity, aquifer/stream recharge, and NOx and non-methane hydrocarbon emissions may also result (Archer et al., 2001). The scientific community will be uniquely challenged to address the ramifications of these looming issues in land use. We believe that linked remote sensing-modeling approaches will be a critical underpinning for the types of landscape and regional monitoring and assessments that will be required by policy makers seeking to make informed decisions.

Acknowledgments. This research was supported by NASA EOS and LCLUC programs. Partial support was provided by the National Science Foundation and the USDA CSRS Rangeland Research Program and National Research Initiative.
6 References


Norris, M. 2000. Biogeochemical consequences of land cover change in eastern Kansas. Kansas State University, Manhattan, KS.


Simmons, M. T. 2003. Tree-grass and tree-tree interactions in a temperate savanna. Ph.D. Texas A&M University, College Station.


CHAPTER 12

ARID LAND AGRICULTURE IN NORTHEASTERN SYRIA
Will this be a tragedy of the commons?

FRANK HOLE\textsuperscript{1}, RONALD SMITH\textsuperscript{2}

\textsuperscript{1}Yale University, Anthropology, New Haven, CT 06520 USA
\textsuperscript{2}Yale University, Geology & Geophysics, New Haven, CT 06520 USA

Abstract

Land use in the Khabur River drainage of northeastern Syria has changed from open rangeland to an intensely cultivated landscape in less than 100 years. In this semi-arid zone successful agriculture usually requires either supplemental or full irrigation. The drivers of the changes have included settlement of refugees, rapid population growth, ambitious plans to develop the water resources for summer cropping, decisions by individual farmers to install wells, and competing needs and programs in the river's headwaters in Turkey. Collectively these have led to fundamental alteration of the natural drainage in favor of ground water extraction, storage reservoirs and irrigation canals. We have monitored the magnitude of recent changes through satellite imagery. The sustainability of the system under current practices is in doubt.

1 Introduction

A hundred years ago, the semi-arid steppe of northeastern Syria was seasonal grazing land for migratory herders. A few settlements of farmers and agro-pastoralists existed along the perennial rivers, but there were no roads, towns, or government. Today, the Khabur River drainage, encompassing 37,081 km\textsuperscript{2} is one of the most fertile and intensively cultivated regions in the Near East (Figure 1). From a relatively unmanaged landscape a hundred years ago, the region has recently been subjected to water diversion, the extension of agriculture, and settlement of formerly nomadic tribes, that has left no part of the landscape untouched. Although bare of trees today, under natural conditions botanists reconstruct a terebinth-almond-woodland steppe (Moore, et al. 2000:60; Pabot 1957). Unlike most regions of the world, we have archaeological evidence for this region that extends back some 9000 years, as well as sporadic historic documents over the last 4000 years, to help put the trajectory of change into perspective. This case study outlines historical changes and makes use of satellite imagery and historic documents to quantify recent changes. The recent changes have been so rapid and pervasive that they cannot be comprehended from any single locale on the ground. Still less can the local farmers fully comprehend the potential impacts on the land and their livelihoods of the wholesale withdrawal of water for irrigation. While this case study focuses on a single river system, it typifies much that is happening in Southwest Asia generally. In this paper we briefly review the history of land use and use data derived from analysis of satellite images to describe the

magnitude of recent changes. Finally, we consider the sustainability of these developments.

Figure 1. Landsat mosaic of the Khabur River region of northeastern Syria.

2 The Khabur Region

With much of its watershed in southern Turkey, the Khabur River rises in Syria from a series of large karstic springs that provide its principal source (Burdon and Safadi, 1963). The springs ensure a strong perennial flow without high floods, making it possible to irrigate the river bottom by simple gravity flow canals. With an average flow of 43 m$^3$/s (1.4-1.6 x 10$^6$ m$^3$ annual), it is the largest perennial tributary of the Euphrates in Syria and a source of fertile agricultural land within its lower terraces (USDA 1980). The river flows more than 400 km to its confluence with the Euphrates through land that grows progressively drier from north to south. Precipitation at the headwaters in Turkey may reach 800 mm, but at the river's source at Ras al Ain in Syria, precipitation averages some 350-400mm, and then progressively decreases to about 150mm at the Euphrates.

The Khabur region is part of the north Mesopotamian Jazirah, the semi-arid steppe between the Tigris and Euphrates rivers. The climate is a modified Mediterranean pattern with all precipitation occurring during the cool winter months and the remainder of the year is hot and dry. The northern Jazirah has fertile soils and sufficient precipitation to support rain-fed agriculture, but within 50 km to the south of the Turkish border subsistence agriculture is problematic and the situation grows progressively worse the farther south one goes. Interannual variability in precipitation can exceed 100% and drought years are not uncommon so that crops benefit from
irrigation even in the wetter regions and it is indispensable in the drier areas. At Hassakah precipitation was 508.3mm in 1963, but only 95.5mm in 1973 (Sanlaville 1990b). Because of the climate, traditional agriculture depended on fall-sown cereals that ripen in late spring, but with the advent of irrigation two cropping seasons has become the norm, with grain in the winter and cotton or another water intensive cash crop in summer.

The Khabur is home to diverse ethnic and religious groups and the overall population has increased dramatically since the 1950s. During the last years of the French mandate (1940s), the principal towns of Kamishli and Hassakah each had fewer than 15,000 residents, according to the World Gazetteer. In 1960, Hassakah had 19,000 and today has 203,000 residents, while Kamishli has risen from 34,000 in 1960 to 182,000 residents today. To these totals we must add the smaller towns and thousands of villages, hamlets and farmsteads that dot the countryside, for a total that must exceed half a million human residents and an unknown but similarly large number of sheep.

3 Archaeological History

The archaeological history of the Khabur is one of relatively brief episodes of settlement, each lasting several hundred years, separated by a thousand or more years when settlement was sparse (Hole 1997; Hole 2000; Hole 2002). Changes in precipitation, as well as pervasive political conflict account for this uneven history, but modern technology has introduced a new source of potential instability whose outcome is uncertain but troubling (Beaumont 1996). Through massive alteration and exploitation of the natural sources of water, there is the potential to destroy the long-term viability of a rich, but fragile, landscape. Modern development of water resources and land usage are driven by the needs of the burgeoning local population and the agricultural policies of the State.

When the Near East emerged from the Younger Dryas about 11,000 years ago, the Khabur region had no permanent settlements and the land was frequented only by small bands of hunters. As the climate improved with the Holocene Climatic Optimum, the first agricultural villages, dating back to about 7000 BC, were settled close to the flood plain of the Khabur River. For the next three thousand years the plain was sparsely settled and sites generally remained small and were occupied only intermittently. The first large settlements date to the early fourth millennium BC when sites up to 15 ha in area are known (Hole 2000). During the fourth millennium BC, the largest settlement, Hamoukar, spread out over 100 ha, was in the far eastern part of the plain (Ur 2002), and many other smaller sites were along the Khabur River and its tributaries. Apart from agriculture, the major activity was herding of sheep and goats by people who moved seasonally into the Taurus Mountains to obtain fresh pasture in the summer. The extremely large cluster of sites around Hamoukar may have been occupied by such people, who probably abandoned the region in the summer dry season, as Kurdish tribes did until the border with Turkey was closed.

The pattern of settlement and intensity of land use changed markedly around 2600 BC when a number of towns and many small hamlets filled up the landscape (Kouchoukos 1998). Some of the larger towns were walled and enclosed palaces, temples and large granaries that were situated on high mounds. Still the greater amount
of land was used for grazing and after 2500 BC, there were no sites on the river itself, for settlement had moved away from the river out onto the semi-arid steppe or on tributary stream of the Khabur. Because of the move to dry areas, we infer that effective precipitation must have been greater, although demand for wool certainly drove the economic expansion and encouraged exploitation of the steppe where agriculture is risky today (Buccellati 1990; Margueron 1991; Hole 1997). In the best agricultural zone in the northern part of the plain, several large cities developed. The prosperity lasted for only about 300 years and it was followed by a prolonged period of intense aridity starting around 2200 BC that resulted in the abandonment of nearly all sites (Weiss 1997). Not until the modern era was settlement again as widespread and intensive as in the last half of the third millennium BC.

4 Written History

Written history for the region begins in the second millennium BC when herds from the city of Mari on the Euphrates were pastured in the upper Khabur and part of the river was diverted into a canal that brought irrigation water to the edge of the Euphrates and also served for barge transportation (Durand 1990a; Margueron 1990). To divert the water from the river, dams of earth reinforced with bales of reeds held by rocks were built in the river (Finet 1990). Some agriculture was carried out along the river itself and texts report ravages by locusts, as well as flooding that periodically devastated crops (Van Liere 1963).

During the first millennium, the city of Guzana, (modern Tell Halaf), capital of an Aramean kingdom was established at the head of the Khabur. Assyrian forts and settlements were implanted in the Khabur including the large site of Dur Katlimu near Sheddade on the lower Khabur. During Assyrian times, two lengthy canals were built, one on each side of the river, to take water toward the city of Mari on the Euphrates (Durand 1990b; Sanlaville 1990a). The Assyrian presence was largely south of Hassakah until the mid 8th century when they conquered the whole of north Mesopotamia. The upper Khabur district was occupied by transhumant herders who seasonally pastured their sheep on the vast semi-arid steppe.

During the Roman period, the major settlement, Circesium, was at the confluence of the Khabur and the Euphrates and a series of small forts and towns outlined the Roman frontier *limes* (frontier forts) in the northern Jazirah. Archaeological survey has documented the *limes*, and a few of the larger sites on the river (Oates and Oates, 1990; Poidebard 1934), but settlements are rare away from the river which, at this time was said to be navigable.

By 640 AD the entire Jazirah came under Arab control when their armies defeated the Byzantine forces (Glubb 1963). Islamic sites are numerous in the Khabur, but only on the lower river itself was there a city; other sites were uniformly small, generally situated on small, spring-fed wadis or on land with a shallow water table where wells could be dug. Islamic farmers and agro-pastoralists used gravity-flow irrigation in the river flood plain to cultivate summer crops, and established a continuous agricultural presence along the river.

In 1248 AD the Mongols invaded the Khabur region, destroying local settlements and irrigation canals and replacing Arab authority with Turkomen and Kurdish tribes (Boghossian 1952). In the 14th century, the great Arab geographer, "Ibn
Batutah found the district already waste and desolate" as a result of tribal depredations (Epstein 1940), a phrase echoed as late as 1880 "no villages, no cities, only tents and ruins" (Sachau 1883:296).

From the 16th until the 20th century, the Khabur was an administrative district controlled by Ottoman governors, although incursions of diverse refugees and Arab tribes often led to chaotic conditions (Pascual 1980). When there was peace, wheat and barley farmers settled much of the present dry-farming area well south of the current Turkish border. In the 16th century there were only two villages on the entire lower Khabur (south of the Jebel Abd al-Aziz, a low mountain ridge), but nomadic tribes occupied the region seasonally and, in some cases, used water wheels and short canals to irrigate millet, a summer crop (Hütteroth 1992). These tribal canals must have been on the floodplain, unlike the long 3rd millennium Durin canal that is on the east bank (Hütteroth 1990). Kurdish tribes circulated seasonally between the upper Khabur, north of the Jebel abd al-Aziz and pastures in the mountains of southern Turkey.

In 1850, the famous English diplomat-archaeologist, Austen Henry Layard described the middle Khabur. "The richness of its pastures, the beauty of its flowers, its jungles overflowing with animals of all species, its huge trees providing delicious shade against the brilliant sun, all form a terrestrial paradise" (Layard 1853:235). While there must be some exaggeration in this description, in the 19th century, the Khabur was still essentially "virgin" land. To exploit this potential, in 1877 some 15,000 Circassians settled near Ras al Ain after being expelled from Russia, but within twenty years only 200 families remained, the others having been driven out by the Kurds or by malaria that was pervasive in the region (Dodge 1940; Rowlands 1947).

5 French Mandate

Conditions were chaotic because of ethnic and tribal conflict from the turn of the twentieth century through the First World War, when the Ottomans were finally driven out, until the French Mandate that began in 1920. The French had first to conquer the tribes and then begin a process of pacification and settlement in villages. The principal towns of Kamishli and Hassekah were established as garrisons by the French who finally gained control over the northeastern Khabur in 1930 (Boghossian 1952). These towns also sheltered thousands of Christian and Jewish refugees from Turkey who became shop keepers, merchants and artisans (Sarrage 1935; Epstein 1940).

The 1920-30s saw an influx of people, including Armenian, Assyrian and Kurdish refugees, fleeing oppression in Iraq and Turkey (Boghossian 1952). Along with the Kurdish tribes these groups occupied the best agricultural land along the Turkish border and along the upper Khabur River (Dodge 1940). Extensive seeps of petroleum, discovered near the Tigris River as early as 1908, now produce about 80% of Syria's oil and gas. To move oil and agricultural products, a railroad between Aleppo and Kamishli was completed by 1928, and extended to Mosul in Iraq by 1940. The railroad effectively opened the region to development for the first time. However the pace of change was slow as rebellions against the French and between Christians and Arabs rocked the region during the twenties and thirties. As late as the 1940s, "in summer the steppes may be afire, ruining the grazing for the flocks; in winter, when the rainfall is heavy, the route's defiles become a well-nigh impassable morass of mud." Moreover, the cereal crops "are cut primitively by sickle and threshed by animal labor on the
communal threshing-floors of every village. The cultivation is equally primitive: two oxen dragging light ploughs over the fields, barely scratching the surface, and rotation of crops is practically unknown. However, under the Government monopoly of cereal purchases during the recent war, combine-harvesters could be seen plying the fields next to the hand reapers, followed by gleaners" (Rowlands 1947, p. 147-149).

6 Syrian State

The region prospered during the Second World War when the value of agricultural crops, especially that of cotton, rose. In 1950, 17,500 people lived in villages clustered along the upper Khabur River and irrigated 40,300 ha of land, mostly for rice and cotton. But the 50s are more notable for the rapid and vast extension of cultivation into the semi-arid steppe when urban entrepreneurs invested in tractors to cultivate large tracts of formerly grazing land. By the late 40s there were some 400 tractors and 350 combine harvesters in the Jazirah (the steppe between the Euphrates and Tigris rivers) with 715,000 ha under cultivation. In 1961 there were 1241 tractors and 679 combines that worked 1,100,000 ha. By this time the Jazirah had become the breadbasket of Syria (FAO 1966). Under the new Syrian government, several changes in land policy were implemented to divide former large holdings and to exploit formerly uncultivated tribal land. The steppe could be claimed by plowing it and enormous areas of marginal land were brought under speculative cultivation, mostly funded by urban entrepreneurs (Métral 1980). Further, agricultural cooperatives that lent money for tractors and pumps, and subsidies for crops encouraged maximum production (Manners and Sagafi-Nejad, 1985).

Motor pumps became the principal means of extracting water by the early 50s, and freed farmers from riverside agriculture and enabled them to cultivate the entire stretch of bottomland and lower terraces along the Khabur from its headwaters to the Euphrates. In Hassakah province 258 permits for wells had been granted by 1963 and the area of cotton cultivation expanded from 140 ha in 1948 to 150,000 in 1961 (FAO 1966:334). During this time the human population doubled. In the 1960s, the extent of agriculture in the lower Khabur essentially matched that of the 10th-13th centuries (Kerbe 1987).

The effect of pumping from the river can be seen in the difference between flow at gauging stations Ras al Ain and Tell Tamer. In 1942/43 there was little pumping from the river, whereas between 1954 and 1959 it had become considerable and the flow had dropped some 4 m$^3$ (Burdon and Safadi 1963). Water has always been regarded as a common good to be used freely. There is little incentive to conserve when the cost of irrigation from individual wells is only the cost of extraction and when costs for water drawn from canals is a low annual fee independent of volume or frequency of use.

The last two decades have witnessed the most dramatic changes, from predominantly riverside irrigation and vast dry-farmed fields in the semi-arid steppe, to the development of reservoirs and canals feeding planned irrigation districts, to the thousands of isolated farmsteads utilizing ground water far from the river, and the expanding needs of the human population in cities and countryside. With startling rapidity, the flow of the Khabur diminished until it is now dry during the summer, thus precluding cultivation of cotton and other summer crops. Years before, the Jaghjagh,
the eastern tributary of the Khabur, with about half its annual flow, suffered the same fate owing to the extraction of water for irrigation in its upper stretches. These perennial rivers had supplied fish to local markets, as well as water for domestic and agricultural needs as long as people had lived in the region.

7 “FiveYear” Plans

Since the founding of the Syrian State in 1946 at the end of the French Mandate, there have been several studies of water resources and agricultural potential of the region, leading to plans to build dams and reservoirs and to irrigate new lands (USDA 1980) (Kerbe 1987). Some of these Five Year Plans have been implemented; others have not or have been abandoned as headwater developments have usurped the sources of water. For example, the large irrigation district for the lower Khabur that was to have irrigated the land from Sheddade to the Euphrates, has been abandoned in favor of upstream systems, an outcome that was anticipated by an early assessment, "sur le Khabour inférieur, différents sites de barrages ont étudiés, mais semblent techniquement difficiles" (FAO 1966:48). The reservoir to serve this system would have held a billion m$^3$ (Burdon and Safadi 1963:64). Another planned reservoir, El-Feidate, was thought to be capable of holding 440,000 m$^3$ (USDA 1980:I-131,II-35). Today this is just a shallow depression filled by winter rains and fed by a small fresh water spring (Mortier and Safadi, 1967).

An FAO-sponsored study described how a diversion weir a short distance below the springs could provide for summer irrigation of 55-60,000 ha through the use of simple gravity canals on the north bank of the river. It was further estimated that pumping from the river could irrigate 16,000 ha on the south bank. The authors calculated that if water was provided at the rate of 0.5 liters per second per hectare, the demand for 60,000 ha would be 30 cubic meters a second, somewhat less than the minimum monthly flow of the river. If that plan were implemented, in most years there would still be sufficient water to use pumps for the fields on the south bank. Indeed it was estimated that heavy extraction of water might increase discharge by decreasing back-pressure on the natural discharge (Burdon and Safadi, 1963). The residual volume of water stored in the aquifers is estimated to be some 7,415 million m$^3$, "equivalent to six years discharge at the annual average rate" (Burdon and Safadi, 1963:92).

Three major reservoirs have been completed and are operating. The 7th April and 8th March dams northwest of Hasseke impound some 323 million m$^3$, to provide power and irrigation water. The Martyr Basil al-Asad dam below Hassakah is to impound some 605 million m$^3$ and irrigate some 50,000 ha on either side of the river. There are also a number of smaller reservoirs, including one near the karstic springs that was filled to its capacity of 1,900,000 m$^3$ in 1978. Two reservoirs with combined capacity of 21 million m$^3$ were built in the north central sector (USDA 1980) II-35. Three separate sets of reservoirs were also planned in the Kamishli region of northeastern Syria, near the Tigris River. One northeast of the town of Malkieh had a capacity of 50 million m$^3$ (USDA 1980:I-180); another northwest was filled to its capacity of 31 million m$^3$ in 1978 (USDA 1980:I-176); and two small reservoirs were planned near the Tigris River to hold some 8.5 million m$^3$ (USDA 1980:I-179, II-35). All of the reservoirs are shallow and with high evaporation and use for summer irrigation, water levels drop rapidly and expose much bare soil to wind erosion.
The most radical plan for tapping the Khabur was to "drill a large-diameter borehole - to tap the aquifers and concentrate all the waters at one controlled point of discharge...It is realized that complete control could never be established, but if the full annual discharge of Ras-el-Ain could be concentrated into the four irrigation months, with nil discharge during the remaining eight months, then its waters alone could irrigate some 230,000 hectares" (Burdon and Safadi, 1963:93). This plan has, in part been implemented through storing the water in the two large reservoirs (USDA 1980:I-137, II-33).

The reservoirs have also removed a considerable amount of fertile land that was previously cultivated. Since these reservoirs all depend ultimately on surface run-off which supplies the springs, their success depends on upstream diversion of catchments in both Turkey and Syria, as well as on annual precipitation (Burdon and Safadi, 1963). Much of this is out of the control of local authorities in Syria who, in any case, may find it difficult to comprehend the interconnection of factors that impact the water resources. For example, water managers cannot readily know, let alone influence, water extraction in Turkey. Neither can they readily monitor water extraction in Syria from the thousands of individual wells. And they cannot easily determine either the extent or quality of agriculture season by season. The unique perspective afforded by satellite imagery has the potential, therefore, to become an indispensable management tool (Beaumont 1996).

8 Recent Land Use Changes

While we have experience in the Khabur on the ground and have witnessed some of the changes over the last 15 years, we do not have access to up-to-date figures on implementation and use of the various irrigation schemes.

Figures 2a, b. Landsat images of the lower Khabur River (upper right) at its confluence with the Euphrates (lower left) in (a) September 1990 and (b) September 2000. Summer cultivation in the lower stretch of the Khabur is absent in 2000. Note also the increase in Aeolian dust streaks in
2000, caused by the Shamal wind during two dry years, 1999 and 2000. (See CD for color images.)

The satellite images give us our best contemporary overview as well as a time series that we can use to quantify changes in cropping patterns and especially in summer irrigation. For illustration we show satellite images of two years of summer irrigation, 1990 and 2000. A detail of the lower stretch of the river as it enters the Euphrates is particularly revealing (Figure 2). In 1990 there was continuous irrigated agriculture, whereas in 2000 it was nearly absent. It is noteworthy that this stretch of the river is the one that was once scheduled to be irrigated by the reservoir near Sheddade. A few years before, the river was flowing and farmers used diesel pumps to irrigate their fields, and provide their domestic water. Today we see a dry river and unused paddies for summer crops. Farther upstream Figure 3 shows the newly irrigated fields below the new Martyr Basil al-Asad reservoir south of Hassakah. This pair of images also shows how land use has changed from very large cultivated barley fields to small irrigated plots.

Figures 3a, b. Agricultural patterns at the location of the Martyr Basil al-Asad dam on the Khabur River. (a) In September 1990 the entire river flood plain is cultivated and some irrigated fields lay to the east. The patterns of very large dry-farmed barley fields are also seen. (b) In 2000 the barley fields are no longer under cultivation, the reservoir is filled, the floodplain below the dam is sparsely cultivated, and irrigation agriculture has expanded on the steppe.

This type of land use change can be quantified on a much larger scale using a time series of AVHRR image datasets. Of particular interest is the relationship between the major Euphrates and its tributary Khabur River. In Figure 4, we identify fifteen areas of intense summer irrigation in northern Syria and southeastern Turkey. This diagram is derived from August values of Normalized Difference Vegetation Index. In late summer, all dense vegetation is associated with intense irrigation. With the exception of the regions marked Hatay and Adana, all the circled regions draw water from the Euphrates or its tributaries.

Time series of irrigated area in the Khabur basin are shown in Figure 5a, for the period 1981 until 1998. The border area (C), and the northern and central Khabur
floodplains, show dramatic increases, nearly doubling in each case. The southern Khabur, while fluctuating slightly, ends up no larger than it started seventeen years before. This unique “flat” signal is evidence, we believe, of upstream water use and downstream water shortage. More recent data (not shown) shows a dramatic reduction in the lower Khabur summer irrigation between 1998 and 2001. This reduction is the combined result of three dry years and the extra demand of filling the Sheddade reservoir upstream.

Figure 4. AVHRR image of northern Syria showing fifteen regions of intense summer irrigation, detected using the Normalized Difference Vegetation Index. All but Hatay and Adana belong to the Euphrates and its tributaries. Time series for eight of these regions are shown in Figure 5.

Figure 5a.
Figures 5a, b. Time series of flood plain agriculture in the Euphrates and its Syrian tributaries; the Balikh and Khabur. Area of irrigated agriculture (km$^2$) is estimated from AVHRR data, downscaled using less frequent Landsat images. (a) Four sections of the Khabur watershed, (b) Four sections of the Euphrates watershed. Seven of these regions, all but the lower Khabur, show large increases over the period from 1981 to 1998.

The time series of floodplain irrigation along the main Euphrates is also of interest (Figure 5b). Large steady increases are seen in all four regions shown: the Balikh, and the Euphrates south of Raqqa, north of Deir Ezzor and south of Deir Ezzor. In total, the irrigated area in these four regions rose from about 600 km$^2$ to 1400 km$^2$; more than doubling in seventeen years. The impact of this increase is easy to appreciate. If a single cotton or vegetable crop grown in the hot summer sun requires a meter of water depth, the 1400 km$^2$ irrigated area demands a volume of $1.4 \times 10^9$ cubic meters of water.

For comparison, the annual Euphrates discharge is about $3.5 \times 10^9$ cubic meters per year. With even larger increases in irrigation seen upstream in Turkey and downstream in Iraq (not shown), the amount of Euphrates flow remaining for the Amara Marsh in southern Iraq (i.e. the so-called “Garden of Eden”) and the Persian Gulf delta is clearly threatened in years of naturally low discharge.

9 Concluding Remarks

The drivers of land use change in the Khabur are many, some planned by the State but others are the result of individual decisions made for economic opportunity. The socialist government created a series of Five Year Plans for overall economic and social development. From the standpoint of land use the most important of these was to increase agricultural productivity to accommodate a rapidly growing and increasingly affluent population. These plans gave rise to land reform, creation of agricultural cooperatives, economic incentives and subsidies for production, the building of reservoir and canal systems, grain silos and a first class road system. While production has never met the ambitious goals set in these plans, most of the potentially productive land has now been transformed. The result is a landscape that is changing, perhaps irreversibly, as it strains to sustain its viability under unprecedented stresses. Should
the systems now in place fail, either through unwise overexploitation, or because of prolonged natural changes in precipitation, the human toll will be great and may lead to further internal displacement of populations, with predictable potential for conflict.

Beaumont anticipated problems that might arise in the Khabur as a result of the importation of water from the Ataturk Dam in Turkey to support irrigation in the Turkish upper Khabur agricultural district (Beaumont 1996). He reckoned that within twenty years when ground water equilibrium had been established on these newly irrigated lands, the runoff from these fields might double the flow of the river in Syria. In such a case the Khabur might become merely a drain, laden with salt and fertilizer. In a worst case this would destroy agriculture in the Syrian Khabur and perhaps impact the Euphrates through increased salinization and eutrophication. The situation, as monitored by satellite, has not reached that point but the future remains in doubt.

In the arid Near East, flexibility has traditionally provided sustainability. People practicing subsistence agriculture and animal husbandry had sufficient "vacant" land to draw upon in emergency, or they retained mobility to travel temporarily to better pastures. Under present circumstances, there is no "free" land to use or move to so that the principal coping mechanisms for a family are to tighten their belts and seek wage labor. For the government and entrepreneurs, it is to intensify production through the building of reservoirs and canals, the drilling of wells, and eliminating fallow while planting in both winter and summer. The inevitable degradation of productivity may be deferred, however, by use of more efficient methods that use less water (FAO 1966). These measures may alleviate the problem today, but continued intensive use of these fragile lands may ultimately perpetuate and perhaps worsen the degradation.

The question now facing authorities is how to manage a fragile system that may be running at (or worse, above) capacity if and when some parts fail. The problem is worsened by the fact that the ultimate sources of river and ground water lie in Turkey where intensive exploitation of these sources impacts both the quantity and quality of water that enters Syria. Thus, the residents of the Khabur in Syria are at the mercy of their own manipulations, those of the Turks, and of the vagaries of weather. It is not clear that there is either the technical knowledge or the political will to manage these elements toward the end of sustainability for all parties.

From the perspective of space we have a unique ability to monitor changes across the entire watershed. The sources of the Khabur River, long regarded as a "common" for all to use have been usurped by diverting their natural flow and, if Beaumont's predictions hold, augmented with "unnatural" flow from Turkey whose quality and effects could ultimately affect many more than just those already displaced along the former river channel. As we watch the inexorable changes we wonder whether another "tragedy of the commons" may be taking place.

Acknowledgments. This research was supported by NASA NAG5-9316. Benjamin Zaitchik carried out the image analysis in the Center for Earth Observation. A Digital Atlas of photographs of the region can be seen at www.yale.edu.ceo/swap.
10 References


CHAPTER 13

CHANGES IN LAND COVER AND LAND USE IN THE PEARL RIVER DELTA, CHINA

KAREN C. SETO¹, CURTIS E. WOODCOCK², ROBERT K. KAUFMANN³

¹Center for Environmental Science and Policy, Institute for International Studies, Stanford University, Stanford, CA 94305-6055 USA
²Department of Geography & Center for Remote Sensing, Boston University, USA
³Department of Geography & Center for Environmental Science and Policy, Boston University, USA

1 Introduction

Over the last two decades, land-use changes in China have been dominated by an urban transformation unprecedented in human history. The Chinese landscape, which for thousands of years was mainly rural, is becoming increasingly urban. Natural ecosystems, farms, rangelands, towns, and villages are being converted into, or enveloped by, extended metropolitan regions. This urban revolution has profound environmental impacts, including local and regional climate change, loss of wildlife habitat and biodiversity, stress on food production systems, and pressure on water resources. Urbanization can also lead to poor housing conditions, inadequate waste disposal, and rapid spread of infectious diseases. Every aspect of the urbanization process, ranging from the provision of social welfare programs to the construction of transportation infrastructure, presents huge environmental and socioeconomic challenges. However, urbanization does not have only negative impacts. Urban development can offer opportunities for concentrated and efficient land use, progress in environmental quality, and resources for solid and waste water treatment. From a socioeconomic perspective, urbanization can also lead to better living conditions and improvements in well-being through wider availability of health care, better education services, access to reliable energy supplies, and advances in sanitation.

The dramatic urban land-use changes in China have been fueled by economic reforms that lead to impressive economic growth through the 1980s and 1990s. During these two decades, global average annual rates of growth ranged between 2 and 3 percent, respectively, while the Chinese economy grew at breathtaking rates of 10 percent and greater (World Bank 2000). The overwhelming success of the reforms has led to urban changes unparalleled anywhere on Earth. According to recent figures, of the 488 major urban areas in the world, nearly one-quarter are located in China. Furthermore, China’s urban population is predicted to increase by more than the total population of the United States in the next 25 years (United Nations 2001).

Urban changes have led to a greater integration of rural and urban economies: farmers adjust crop types to satisfy urban food demand; urban commodity demands affect natural resource flows; urban remittances contribute to growth in distant regions. Similarly, physical demarcations between country and city are blurring as areas distant from city centers become absorbed into the sphere of the urban economy. Yet while the physical and economic landscapes have become more intertwined, urban changes are
causing other striking differences in society: cities are populated by a disproportionate number of young people whereas countryside villages house mainly the aged. Many of the industrial and export-oriented cities rely on young female migrant workers, who often move hundreds if not thousands of miles from their hometowns, further biasing the local sex ratio in a society with a strong preference for sons. The family structure also is changing, with fewer extended members residing in a single household.

The effects of urban land-use change on socioeconomic and ecological systems are complex, as are the factors that influence urban land use. Land-use change, and in particular urban change, is a significant component of global environmental change, and yet the drivers of these changes are not well understood. Therefore, monitoring the spatial and temporal patterns of land use, and identifying the key variables that influence land-use changes are critical elements of global change research. This chapter synthesizes research in the Pearl River Delta that focused on identifying patterns of land-use change, developing empirical models of the macro-level drivers of these changes, and evaluating the social and ecological impacts for the period 1988 to 1996.

2 The Pearl River Delta

The Pearl River (Zhujiang) Delta (PRD) is one of the most economically vibrant regions in China (Figure 1). Located in the southern province of Guangdong, the climate of the Delta is characterized by a dry season from November through April, and a rainy season from May through October. The summers are hot and humid, and temperatures in July and August sometimes soar to 38°C. The Delta’s 21 million residents comprise one-third of the total population of the province, and the region has a population density of 850 persons per square kilometer, which is more than twice the provincial average.

Figure 1. The Pearl River Delta, Guangdong Province, China

Between 1985 and 1997, urban population in the Delta increased by 68 percent while agricultural population increased by only 9 percent (Statistical Bureau of Guangdong various years). The region has been settled since the Zhou Dynasty (1027-221 B.C.), and has a strong agricultural history of rice and sugarcane production. The rich red
alluvial soils support two to three crops per year, and the natural vegetation is dominated by pines and acacias.

The remarkable economic growth and accompanying changes in land use in the PRD can be attributed to seven major factors. First, the region benefited from national reform policies in the agricultural sector. Beginning in the late 1970s, price reform and the elimination of collective farming in favor of the “household responsibility system” lead to an increase in prices for agricultural outputs and allowed agricultural decisions to be made by the household rather than the commune or production brigade. The net effect was that crop yields increased and a surplus of agricultural workers became available for other sectors of the economy. Second, decentralization policies allowed provincial and local governments more autonomy from the central government to devise and implement their own development strategies. This allowed the creation of different incentive structures and policies to stimulate investment, economic development, and land use.

Third, the establishment of three special economic zones (SEZs) in the 1980s, Shantou, Shenzhen, and Zhuhai, and the formation of the Pearl River Delta Economic Open Region in 1985, helped the Delta to attract foreign investment and transform itself into an export-oriented region (Chu 1998; Lin 1997). After the SEZs and the PRD Open Region were established, Hong Kong entrepreneurs began to move their operations across the border, first into Shenzhen, and eventually throughout the Delta. These overseas ventures had a considerable impact on the pace and structure of economic and urban development (Eng 1997). Fourth, the central government initiated a series of sweeping reforms beginning in 1978 that included the promotion of township and village enterprises (TVEs). TVEs were initially agricultural collectives that produced agricultural-related goods, and evolved into one of the pillars of economic growth. With the availability of surplus rural laborers, TVEs diversified considerably into other sectors and their relative freedom from bureaucratic controls and low labor costs made them attractive partners for foreign investment (Putterman 1997).

Fifth, land reform in 1988 allowed the transfer of land-use rights through negotiation, auction, or bid (Sharkawy et al., 1995). Prior to the land reform, all land in China belonged to the state and was managed by brigades and collectives. One result of the land reform is that individuals and collectives can rent or lease their land to foreign and local ventures. Sixth, before 1978, China had two policies that effectively limited population mobility, especially from rural to urban areas. The household registration system, hukou, determined the residency status of an individual, while the work unit, danwei, was an important provider of basic goods and services such as housing, health care, food ration tickets, and education. Together, the hukou and danwei were powerful forces to control urbanization and internal migration. Reforms have since relaxed the hukou and reduced the importance of the danwei (Mallee 1996; Smart and Smart, 2001). Finally, the role of Hong Kong as a catalyst for economic and urban development in the PRD needs to be underscored. The bulk of foreign direct investments in the PRD (nearly 75 percent in 1996) come from Hong Kong. The geographic proximity and cultural ties to Hong Kong provides the PRD with large investment inflows, access to technological innovations, and managerial acumen.
3 Land Cover and Land Use Changes

Land-cover and land-use change trajectories were developed with Landsat TM data acquired annually from 1988 to 1996. A number of methodologies were tested to assess land-cover and land-use change, and to calibrate the images for sensor, seasonal, and atmospheric differences. The techniques tested and their relative successes are provided elsewhere in the literature and will not be repeated here (Kaufmann and Seto, 2001; Seto et al., 2002; Song et al., 2001). What follows is a discussion of the primary land-cover and land-use changes in the Delta during the period 1988 to 1996, and the key drivers and impacts of these changes.

3.1 CHANGES IN URBAN AREAS

By far, the most dramatic land-use change is the growth of urban areas. During the period 1988 to 1996, nearly 7 percent of the PRD, or 1905 km$^2$, was converted to urban uses from agriculture, natural vegetation, or water. In 1988, there were only 720 km$^2$ of urban land. By 1996, urban areas constituted 10 percent of the PRD, or 2625 km$^2$. Changes in policy gave rise to three periods of urban development during the 1980s and 1990s. The first period, the 1980s, was marked by slow urban growth. Promotion of town and village enterprises (TVEs) during this time turned towns into economic and industrial engines, but this had relatively little effect on extensive urban land-use change. Foreign investments played a part in promoting TVEs and the expansion of towns and villages, but these were on a relatively small scale. The remote sensing analysis includes the tail end of this period (Figure 2).

Figure 2. New urban development by year for selected counties.
The Tiananmen Square incident in 1989 cast a shadow over the investment environment until Deng Xiaopeng’s southern China tour in January 1992 (Figure 3). After his visit to the region and affirmation of the government’s commitment to open door policies, investments more than tripled the following year, ushering in the second period of development: large-scale urban land-use change driven by foreign investment. For some counties such as Dongguan and Shenzhen, most of the urban land-use changes occurred during 1992-1993.

Figure 3. Foreign investments in the Pearl River Delta Open Economic Zone, 1988-1997.

The first two periods of urban development laid the foundation for future growth, with investments centered in the SEZs and the major cities in the Guangzhou-Shenzhen corridor. These periods were characterized by seemingly chaotic patterns of urban expansion, which often were determined by the construction of new transportation networks. In the third phase of urban development, investments moved geographically south of Guangzhou, the provincial capital, and intermediate-sized cities grew rapidly throughout the region (Figure 4). Although little of the development during this phase was planned or regulated, there was less of a boomtown atmosphere than during the previous two periods.
One striking characteristic of urban land-use change in the Delta is the near absence of higher-level regional planning from the provincial or state offices. What appears to be an unorganized mélange of disparate aims is strongly impacted by social-political networks and economic interests. The uncoordinated activities of multiple interests have generated urban land-use change that shifted the influence of major cities. The two primary cities and regional powerhouses were once Guangzhou and Hong Kong, located south of Shenzhen, but their industrial and economic influence has declined over the years. As the greater Delta transforms into a large urban cluster, the economic and political centers of gravity continue to shift in coordination with urban land-use change. For example, Shenzhen, which was once a quiet fishing community, has transformed into a major metropolitan area that rivals Hong Kong (Figure 5).
New urban growth includes highways, residential, commercial, and industrial development, and land cleared for future urban development. Increased incomes also have resulted in greater private vehicle ownership and a higher demand for highways. During the study period, the number of motor vehicles increased by 259 percent and the length of highways increased by two-thirds (Table 1). Express highways crisscross the Delta, facilitating transportation and delivery of goods to markets and ports. Although the remote sensing analysis does not differentiate among types of structures, official statistics suggest that residential housing construction comprises a significant portion of the new urban space. As a result of economic development and rises in income, the quantity of housing stock increased for city residents (Table 2). Between 1985 and 1996, per capita living space for urban residents nearly doubled. This trend is not limited to cities. In general, living space increased for all residents (Table 3). In rural areas, the increase in agricultural wages, the ability to contract out farming, and the leasing of agricultural land all contributed to the growth in rural incomes and an increase by 50 percent in per capita living space between 1985 and 1996. Housing conditions also have improved dramatically, and American-style villas are increasingly more popular (Figure 6).

Table 1. Main indicators of transportation services

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Length of highways (km)</td>
<td>53 820</td>
<td>54 671</td>
<td>84 563</td>
<td>89 631</td>
</tr>
<tr>
<td>Number of motor vehicles</td>
<td>324 195</td>
<td>402 086</td>
<td>1 147 348</td>
<td>1 163 339</td>
</tr>
</tbody>
</table>


Table 2. Average per capita residential floor space for selected cities (m²)

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Guangzhou</td>
<td>8.58</td>
<td>10.08</td>
<td>10.78</td>
</tr>
<tr>
<td>Shenzhen</td>
<td>14.71</td>
<td>14.22</td>
<td>19.27</td>
</tr>
<tr>
<td>Dongguan</td>
<td>15.6</td>
<td>23.61</td>
<td>24.58</td>
</tr>
<tr>
<td>Foshan</td>
<td>10.77</td>
<td>13.9</td>
<td>18.85</td>
</tr>
</tbody>
</table>

Table 3. Average per capita residential floor space of urban/rural residents (m²)

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Rural</td>
<td>14.87</td>
<td>17.39</td>
<td>20.83</td>
<td>22.32</td>
</tr>
</tbody>
</table>

Source: *Statistical Yearbook of Guangdong, 1997*.

Figure 6. New housing development in the Pearl River Delta

![Photo: K. Seto](image)

3.2 CHANGES IN AGRICULTURAL AREAS

The increase in urban areas has been mainly from the conversion of agricultural land. A total of 1392 km² of agricultural land – an area the size of San Francisco or Madrid – were converted to non-agricultural uses during the study period. Four major types of agricultural land conversion have occurred. The construction of industrial centers, residential complexes, and factories has lead to the conversion of agricultural land on a large-scale. On a smaller scale, improvement of houses owned by farmers and agricultural workers also reduces the amount of land available for agriculture. Third, highway development has divided agricultural plots and removed them from cultivation. The fourth type of agricultural land conversion is the flooding of fields for water reservoirs and dams, although this constitutes less than 1 percent of the total study area (16 km²). With increases in water demand by the residential and industrial sectors, reservoirs and dikes have been constructed to provide the booming region with an adequate water supply. A majority of the agricultural plots that have been converted are the most productive—they are in the fertile mouth of the delta.

Remote sensing-based estimates of total agricultural land and agricultural land loss differed from figures reported in statistical yearbooks (Seto et al., 2000). Agricultural land loss from the remote sensing analysis is about 11 percent greater than the figures reported in statistical yearbooks (Figure 7), and total amount of agricultural land is about 115 percent greater than official numbers. The higher estimates of loss of farmland may be in part due to the coarse resolution of the Landsat data which cannot differentiate among irrigation ditches, dirt paths, small houses, and other land uses which co-exist with agriculture. Even with this potential bias, there are reasons to believe that the total amount of agricultural land was systematically underreported by
farmers due to institutional factors such as the tax system and historical grain quotas. Furthermore, variations in agricultural crop production and land markets among counties may explain some of the differences in the figures of farmland conversion. For example, in Dongguan, where lychees are important sources of agricultural income for the county government, restrictions on farmland conversion may provide incentives for underreporting agricultural land loss. Conversely, in the coastal county of Panyu, a government program to reclaim the delta for agricultural production may provide incentives for over reporting of agricultural land loss.

Figure 7. Agricultural land loss for selected counties, 1990-1996

Source: Seto et al., 2000.

The loss of agricultural land has been partly mitigated by this cropland expansion into the mouth of the delta as well as by agricultural intensification through increasing fertilizer use, irrigation, and reducing fallow periods. During the study period, approximately 151 km$^2$ of the Delta’s water areas were converted to farmland, mainly in the western region of the Delta. This increase offsets about 11 percent of the loss in agricultural land converted for urban uses. Total amount of cropland has decreased, but in general, grain yields and market value of agriculture have increased (Table 4). Farmers now produce crops for urban diet, and respond quickly to market signals. The loss of agricultural land has given rise to urban agriculture, which is now commonplace throughout the delta. Urban agriculture is engaged in by locals and by cash-strapped migrants who till roadsides and small agricultural patches as a means to provide food.
Table 4. Sown area, yield, and total output of selected crops for select cities

<table>
<thead>
<tr>
<th>Province</th>
<th>Sown Area</th>
<th>Yield</th>
<th>Total Output</th>
<th>Sown Area</th>
<th>Yield</th>
<th>Total Output</th>
</tr>
</thead>
<tbody>
<tr>
<td>Guangzhou</td>
<td>33,20,100</td>
<td>334</td>
<td>1,108,700</td>
<td>2,698,372</td>
<td>368</td>
<td>992,432</td>
</tr>
<tr>
<td>Shenzhen</td>
<td>281,900</td>
<td>269</td>
<td>75,700</td>
<td>19,448</td>
<td>339</td>
<td>6,586</td>
</tr>
<tr>
<td>Dongguan</td>
<td>1,236,400</td>
<td>373</td>
<td>460,900</td>
<td>626,871</td>
<td>387</td>
<td>242,778</td>
</tr>
<tr>
<td>Jiangmen</td>
<td>4,494,500</td>
<td>298</td>
<td>1,341,500</td>
<td>4,054,284</td>
<td>355</td>
<td>1,440,676</td>
</tr>
<tr>
<td>Foshan</td>
<td>2,200,800</td>
<td>347</td>
<td>762,600</td>
<td>1,253,210</td>
<td>372</td>
<td>465,800</td>
</tr>
<tr>
<td>Zhongshan</td>
<td>1,338,700</td>
<td>397</td>
<td>531,700</td>
<td>734,075</td>
<td>370</td>
<td>271,563</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Province</th>
<th>Sown Area</th>
<th>Yield</th>
<th>Total Output</th>
<th>Sown Area</th>
<th>Yield</th>
<th>Total Output</th>
</tr>
</thead>
<tbody>
<tr>
<td>Guangzhou</td>
<td>3,200,900</td>
<td>340</td>
<td>1,088,700</td>
<td>2,541,573</td>
<td>376</td>
<td>955,456</td>
</tr>
<tr>
<td>Shenzhen</td>
<td>259,900</td>
<td>276</td>
<td>71,600</td>
<td>5,886</td>
<td>306</td>
<td>1,801</td>
</tr>
<tr>
<td>Dongguan</td>
<td>1,151,500</td>
<td>390</td>
<td>448,900</td>
<td>559,499</td>
<td>401</td>
<td>224,469</td>
</tr>
<tr>
<td>Jiangmen</td>
<td>4,148,600</td>
<td>311</td>
<td>1,291,400</td>
<td>3,646,458</td>
<td>375</td>
<td>1,366,894</td>
</tr>
<tr>
<td>Foshan</td>
<td>2,095,600</td>
<td>356</td>
<td>747,000</td>
<td>1,158,925</td>
<td>381</td>
<td>441,859</td>
</tr>
<tr>
<td>Zhongshan</td>
<td>1,293,800</td>
<td>402</td>
<td>520,700</td>
<td>691,090</td>
<td>372</td>
<td>257,085</td>
</tr>
</tbody>
</table>


1 In mu. 15 mu = 1 ha  2 Kilogram per mu  3 Metric Tons

3.3 CHANGES IN NATURAL VEGETATION

The Delta’s long history of agriculture and human settlement suggests that extensive deforestation occurred well before the current period of economic development. Most of the intact large tracts of forests are located in the mountainous regions north of the Delta’s basin. The land-cover changes within the Delta were mainly the conversion of shrubs, patches of forests, and hills. Approximately 529 km² of the Delta were converted from natural vegetation or water to urban uses.

Urban development has lead to the quarrying of hills for construction inputs, which has caused widespread soil erosion. A study of the changes in topography might conclude that the region has become more leveled over the last two decades as a result of quarrying and new construction. Soil erosion has become prevalent in the Delta, and the Chinese Academy of Forestry has engaged in collaborative reforestation projects with international organizations. The ninth five-year plan includes increasing forest coverage by planting 1.2 million hectares of forest in the Delta. Plantation efforts have focused on fast growing species of *Eucalyptus*, *Acacia* and native Chinese pine. Especially prevalent is *Acacia mangium* which is tolerant of poor soils and has grown successfully under similar conditions in other tropical environments (Figure 8).
3.4 DRIVERS OF LAND-USE CHANGE

Land-use change is a complex function of factors that interact at multiple temporal and spatial scales. Results from empirical models suggest that a myriad of factors affect the conversion of cropland and forests, and that there are different underlying dynamics (Seto and Kaufmann, 2003). The ratio of agricultural land productivity and industrial land productivity is one of the key variables that cause the conversion of both natural ecosystems and agricultural land. If the economic return to land used for manufacturing is greater than the economic return to land used for farming, cropland will be converted to industrial uses.

One of the major factors associated with land conversion is investments. Investments from abroad generally fuel large-scale projects while domestic investment sources generally fund small or medium-scale projects. Large projects employ numerous local workers, whose incomes rise as a result. As disposable incomes rise, workers may improve their housing conditions by constructing new homes or refurbishing their existing homes. In many towns, higher incomes increase local demand for shopping arcades, cars, and luxury restaurants. This rise may further provide incentives to build residential and commercial complexes which are funded by overseas investors. Therefore, there is a positive feedback loop between investments and land conversion.

Another reason for the conversion of agricultural land to non-agricultural uses is the difference in farm wages versus fees from leases. When farmers can make more money from renting out or leasing their land, there is very little incentive to keep land under cultivation. Poor economic returns to agricultural land could be in part due to the household responsibility system which divided land into plots based on family size. This fragmentation discourages economies of scale and prevents some farmers from cultivating cash crops that requires a larger continuous tract of cropland.

The analysis suggests that the causes of land-use change have multiple feedbacks that include income growth, consumer demand, and population migration. Higher incomes have fuelled a shift from agricultural to non-agricultural livelihoods. Similarly, township and village governments have been agents of urbanization within their levels of jurisdiction by facilitating development of town village enterprises in the region and by endorsing local efforts to cluster privately-owned firms through the development of industrial estates and technology corridors. The results indicate that
urbanization in the Pearl River Delta has been caused largely by exogenous factors, such as international capital movements, and that local land users do not have much influence over large-scale projects.

3.5 BIOPHYSICAL IMPACTS OF CHANGES

Land-use and land-cover changes in the PRD have affected two key components of the terrestrial carbon cycle, net primary production (NPP) and the carbon stock. Land-use change, and in particular the loss of cropland, caused significant modifications to the regional carbon budget by reducing the annual NPP and the size of the terrestrial carbon reservoir (Dye et al., 2000). Analysis of NPP indicates that terrestrial changes during the study period reduced the amount of annual atmospheric carbon assimilated into phytomass by approximately 1.55 Mt (-7.5%). More than half of this reduction is due to the conversion of agricultural land (Table 5). The 7.5 percent decline in annual NPP is a measure of the reduction in total photosynthetic capacity and carbon sequestration potential of terrestrial ecosystems. The average estimate of annual carbon released due to land-use and land-cover change is $1.3 \times 10^6$ t C, with a total of $11.7 \times 10^6$ t C for the entire study period (Figure 9). While this is not an insignificant amount, it only represents 8 percent of the emissions from fossil fuels for 1996. Given the region’s land use and development trajectory, it is likely that carbon sources from land-use change will continue to be overshadowed by fossil fuel emissions.

Table 5. Effects of Land Use Change on NPP

<table>
<thead>
<tr>
<th>Land Class</th>
<th>NPP (t C ha$^{-1}$ yr$^{-1}$)</th>
<th>Regional Total (NPP$_{tot}$) (10$^4$ t C yr$^{-1}$)</th>
<th>Net Change in NPP$_{tot}$ for 1988-1996 (10$^4$ t C yr$^{-1}$) (% change)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Natural vegetation</td>
<td>9.3</td>
<td>1247.8</td>
<td>-70.6</td>
</tr>
<tr>
<td>Agriculture</td>
<td>7.6</td>
<td>813.5</td>
<td>-84.8</td>
</tr>
<tr>
<td>Total</td>
<td>2061.3</td>
<td>1905.9</td>
<td>-155.4</td>
</tr>
</tbody>
</table>

Source: (Dye et al. 2000).

Figure 9. Comparison of carbon emissions from land-use change and fossil fuels

Adapted from (Dye et al., in review).
4 Conclusions

Human activity has changed the face of Earth, and nowhere is this more evident than in the Pearl River Delta. A transformed economy has changed the physical landscape at a rate and level unprecedented in the history of the country. Much of the land-use change is conversion from agriculture and natural vegetation to urban areas. From an estimated 720 km$^2$ of urban area in 1988, or 2.67 percent of the study area, the Delta’s urban land increased to over 2625 km$^2$ by 1996. Urban areas now comprise almost 10 percent of the Delta. While roughly one-quarter of the new urban areas were previously natural vegetation or water, most were converted from farmland, approximately 1376 km$^2$. One of the most noticeable effects of land-use change is the improvement in living standards and human well-being. Although urban growth in the Delta has been impressive during the study period, the region is in nascent stages of urban development. Because land-use change is closely correlated with investments and economic development, continued land conversion is expected in the next several decades until the region reaches an economic or development plateau. Indeed, in much of China, similar patterns will persist as the country continues to grow economically.

5 References


Section III  Cross Cutting Themes, Impacts and Consequences
CHAPTER 14

THE EFFECTS OF LAND USE AND MANAGEMENT ON THE GLOBAL CARBON CYCLE

R.A. HOUGHTON1, FORTUNAT JOOS2, GREGORY P. ASNER3

1Woods Hole Research Center, Woods Hole, MA 02543 rhoughton@whrc.org
2Climate and Environmental Physics, Physics Institute, University of Bern Sidlerstr. 5, CH-3012 Bern, Switzerland joos@climate.unibe.ch
3Department of Global Ecology, Carnegie Institution, 260 Panama Street, Stanford, CA 94305; gasner@globalecology.stanford.edu

Abstract

Major uncertainties in the global carbon (C) balance and in projections of atmospheric CO2 include the magnitude of the net flux of C between the atmosphere and land and the mechanisms responsible for that flux. A number of approaches, both top-down and bottom-up, have been used to estimate the net terrestrial C flux, but they generally fail to distinguish possible mechanisms. In contrast, calculations of C-fluxes based on land-use statistics yield both an estimate of flux and its attribution, that is, land-use change. A comparison of the flux calculated from land-use change with estimates of the changes in terrestrial C storage defines a residual terrestrial C sink flux of up to 3 PgC yr⁻¹, usually attributed to the enhancement of growth through environmental changes (for example, CO2 fertilization, increased availability of N, climatic change). We explore whether management (generally not considered in analyses of land-use change), instead of environmental changes, might account for the residual sink flux. We are unable to answer the question definitively. Large uncertainties in estimates of terrestrial C fluxes from top-down analyses and land-use statistics prevent any firm conclusion for the tropics. Changes in land use alone might explain the entire terrestrial sink if changes in management practices, not considered in analyses of land-use change, have created a sink in the northern mid-latitudes.

1 Introduction

Several lines of evidence suggest that terrestrial ecosystems have been a net sink for carbon (C) in recent years. The evidence is more compelling for northern mid-latitude lands than it is for the tropics, but a number of analyses suggest a terrestrial C sink in the tropics as well. The mechanisms thought to be responsible for the terrestrial sink have included factors that enhance growth, such as CO2 fertilization, nitrogen deposition, and the differential effects of climate variability on photosynthesis and growth relative to respiration and decay. Changes in land use may also lead to terrestrial C sinks through the regrowth of forests following agricultural abandonment or harvest, but the net effect of land-use change is estimated to have released C globally
and, thus, does not explain the net terrestrial sink. The recent analysis by Caspersen et al. (2000) suggests that 98% of the C accumulation in trees in five eastern U.S. states can be explained by the age structure of the forests (that is, regrowth), and only 2% may be attributed to enhanced growth. If environmental growth enhancement were negligible, it would have important implications for the future of the terrestrial C-sink. We explore in this paper whether the results from analyses of land-use change are consistent with other observationally-based estimates of the terrestrial C fluxes. The review is similar to a recent comparison of methods used to estimate terrestrial sources and sinks of C (House et al., in press), but the emphasis here is on changes in land use.

2 Methods for Evaluation of the Terrestrial Flux of Carbon

Both top-down approaches based on atmospheric observations and bottom-up approaches based on terrestrial data have been used to estimate C-fluxes between terrestrial ecosystems and the atmosphere. In global top-down approaches, a basic assumption is that the change in atmospheric, oceanic and terrestrial C storage equals fossil C emissions. Fossil emissions are known from trade statistics (Marland et al., 2000) and the atmospheric change from direct atmospheric measurements or from ice core data. The partitioning between the terrestrial and oceanic sink is determined from additional information, such as the observed trend in atmospheric O$_2$ or $^{13}$CO$_2$. The global budget equations for atmospheric CO$_2$ and O$_2$ (Keeling et al., 1996; Battle et al., 2000) or for CO$_2$ and $^{13}$CO$_2$ (Joos and Bruno, 1998; Keeling et al., 2001) are solved for changes in terrestrial and oceanic C storage. The recent IPCC assessment (Prentice et al., 2001) reported an average net sink during the 1990s of 1.4 ($\pm$0.7) PgC yr$^{-1}$ for the world’s terrestrial ecosystems. In other global methods, the oceanic sink is estimated from oceanic tracer distributions (Quay et al., 1992; Heimann and Meier-Reimer, 1996; Gruber and Keeling, 2001; Takahashi et al., 2002; McNeil et al, 2003) or by applying an ocean model (Siegenthaler and Oeschger, 1987; Bruno and Joos, 1997). Then, the change in terrestrial storage is calculated by difference from the atmospheric C balance.

In a second top-down approach, regional surface C fluxes are estimated from the observed spatial gradients in the atmospheric concentrations of CO$_2$ (sometimes including the distribution of $^{13}$CO$_2$ and global O$_2$ as well) in combination with an atmospheric transport model (Rayner et al., 1999; Bousquet et al., 2000; Guerney et al., 2002). Results of this regional inverse approach depend on the transport model and the way boundary conditions are prescribed (Guerney et al., 2002). Globally, the world’s terrestrial regions summed to a net sink of 1.4 ($\pm$0.8) PgC yr$^{-1}$ for the period 1992-1996 (Gurney et al., 2002), identical to the average obtained from global approaches (Prentice et al., 2001).

An important distinction exists between global and regional inverse approaches. In the global top-down approaches, changes in C storage, that is in the oceanic and terrestrial C sink, are calculated. In contrast, the regional inverse method yields C-fluxes between the land or ocean surface and the atmosphere. These C-fluxes include both natural and anthropogenic components. Horizontal exchange between regions must be taken into account to estimate regional and global changes in oceanic and terrestrial storage. For example, the fluxes will not accurately reflect changes in the amount of C on land or in the sea if some of the C fixed by terrestrial plants is
transported by rivers to the ocean and respired there (Sarmiento and Sundquist, 1992; Tans et al., 1995; Aumont et al., 2001).

Bottom-up approaches include data from forest inventories (Goodale et al., 2002) and analyses of land-use change (Houghton, in press). Forest inventories provide systematic measurement of wood volumes from more than a million plots throughout the temperate and boreal zones. Converting volumes to total biomass and accounting for the fate of harvested products and changes in the pools of woody debris, forest floor, and soils yield C budgets for the forests of northern mid-latitudes. A net sink of 0.6-0.7 PgC yr$^{-1}$ was recently reported for the northern mid-latitude forests (Goodale et al., 2002). Unfortunately, forest inventories are rare over large regions of the tropics, and similar comparisons are not possible there. However, long-term measurements on a small number of permanent plots in tropical forests suggest that undisturbed forests may be functioning as a large C sink (Phillips et al., 1998). This result is discussed in more detail below.

A second bottom-up approach estimates the flux of C associated with changes in land use. These analyses are based on rates of land-use change and the changes in C storage (in living and dead vegetation, soils, and wood products) that accompany a change in land use. The net effect of deforestation, reforestation, cultivation, and logging is calculated to have released an average of 2.0 PgC yr$^{-1}$ globally during the 1980s, and 2.2 PgC yr$^{-1}$ during the 1990s (Houghton, in press). The approach does not consider all lands, but only those that have been cleared, cultivated, planted, logged, and, in some analyses, burned. The calculated fluxes include the C sinks associated with forest (re)growth as well as the sources from burning and decay of organic matter.

3 Do Changes in Land Use Explain Net Terrestrial Sources and Sinks of Carbon?

The source of C from land-use change, compared to the net sink found by other analyses, suggests that other factors are important for explaining the sink. However, most of the sink estimates need adjustments to make them comparable. Furthermore, a number of recent analyses have modified estimates of the magnitude of the terrestrial flux. In this paper we review these new estimates from the perspective of land-use change. In particular, do changes in land use and land management account for the entire change in C storage in terrestrial ecosystems? Or do other factors contribute significantly to a terrestrial sink?

3.1 THE CURRENT (1990s) GLOBAL CARBON BALANCE

According to the latest IPCC assessment, terrestrial ecosystems, globally, were a net sink for C, averaging 0.2 ($\pm 0.7$) PgC yr$^{-1}$ in the 1980s and 1.4 ($\pm 0.7$) PgC yr$^{-1}$ for the 1990s (Prentice et al., 2001). The reason for the large increase between the 1980s and 1990s is unknown. Also unexplained is the apparent decrease in the net oceanic C sink from the 1980s to the 1990s. Given the larger emissions from fossil fuels and the higher atmospheric concentration of CO$_2$ in the second decade, one would have expected the oceans to take up more, not less, C. The partitioning of the C sink between land and ocean was based on recent atmospheric CO$_2$ and O$_2$ data and included a small correction for the outgassing of O$_2$ from the oceans (Prentice et al., 2001).
More recent analyses (Bopp et al., 2002; Keeling and Garcia, 2002; Plattner et al., 2002) determined that the outgassing of $O_2$ was much larger than estimated by Prentice et al. (2001). The recalculated partitioning of C uptake between land and sea (Table 1) by Plattner et al. (2002) shows a larger oceanic uptake in the 1990s than the 1980s, in line with results of oceanic models. The terrestrial uptake is more similar between decades than estimated by Prentice et al. (2001) (an average difference of 0.3, rather than 1.2, PgC yr$^{-1}$). The net terrestrial flux averaged 0.4 and 0.7 PgC yr$^{-1}$ during the 1980s and 1990s, respectively.

Table 1. Global C budgets for the 1980s and 1990s (PgC yr$^{-1}$). Negative values indicate a withdrawal of CO$_2$ from the atmosphere.

<table>
<thead>
<tr>
<th></th>
<th>1980s</th>
<th>1990s</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fossil fuel emissions*</td>
<td>5.4 ± 0.3</td>
<td>6.3 ± 0.4</td>
</tr>
<tr>
<td>Atmospheric increase*</td>
<td>3.3 ± 0.1</td>
<td>3.2 ± 0.2</td>
</tr>
<tr>
<td>Oceanic uptake**</td>
<td>-1.7 ± 0.6</td>
<td>-2.4 ± 0.7</td>
</tr>
<tr>
<td>Net terrestrial sink**</td>
<td>-0.4 ± 0.7</td>
<td>-0.7 ± 0.8</td>
</tr>
<tr>
<td>Land-use change***</td>
<td>2.0 ± 0.8</td>
<td>2.2 ± 0.8</td>
</tr>
<tr>
<td>Residual ‘terrestrial’ sink</td>
<td>-2.4 ± 1.1</td>
<td>-2.9 ± 1.1</td>
</tr>
</tbody>
</table>

* from Prentice et al. (2001)
** from Plattner et al. (2002)
*** from Houghton (in press)

Estimates of a global net C flux from the atmosphere to the land, calculated by summing regional fluxes, which, in turn, are obtained by inverting the spatial CO$_2$ distribution, average around 1.4 PgC yr$^{-1}$ (Gurney et al., 2002), higher than that obtained from changes in O$_2$ and CO$_2$ (0.7 PgC yr$^{-1}$). However, these estimates of flux need to be adjusted for riverine transport to obtain an estimate of the terrestrial C sink. Several studies have tried to adjust atmospherically-based C fluxes to account for this transport of C by rivers. Sarmiento and Sundquist (1992) estimated a pre-industrial net export by rivers of 0.4-0.7 PgC yr$^{-1}$, balanced by a net terrestrial uptake of C through photosynthesis and weathering. Aumont et al. (2001) estimated a terrestrial uptake due to continental weathering of 0.7 PgC yr$^{-1}$. Reducing the net terrestrial sink obtained through inverse calculations (1.4 PgC yr$^{-1}$) by 0.6 PgC yr$^{-1}$ yields a result of 0.8 PgC yr$^{-1}$, overlapping with the estimate obtained though changes in O$_2$ and CO$_2$ concentrations (Table 2). The two methods based on atmospheric measurements yield similar global estimates of a small net terrestrial C sink. The source of 2.2 PgC yr$^{-1}$ calculated from changes in land use is very different from this global net terrestrial sink (0.7 PgC yr$^{-1}$).

3.2 A RESIDUAL (TERRESTRIAL) FLUX OF CARBON

If the net terrestrial flux of C during the 1990s was 0.7 PgC yr$^{-1}$, and 2.2 PgC yr$^{-1}$ were emitted as a result of changes in land use, then 2.9 PgC yr$^{-1}$ must have accumulated on land for reasons not related to land-use change. This gross sink is called the residual
terrestrial sink (Table 1) (formerly called the “missing sink”). The C released from
land-use change and the residual sink sum to the observed net sink. To the extent that
the residual terrestrial sink exists at all suggests that processes other than land-use
change are affecting C storage on land. On the other hand, the residual sink is
calculated by difference; if the emissions from land-use change are overestimated, the
residual sink will also be high.

Table 2. Estimates of the annual terrestrial C sink (PgC yr\(^{-1}\)) in the 1990s according to different
methods. Negative values indicate a terrestrial sink.

<table>
<thead>
<tr>
<th></th>
<th>O(_2) and CO(_2)</th>
<th>Inverse calculations</th>
<th>Forest inventories</th>
<th>Land use change</th>
</tr>
</thead>
<tbody>
<tr>
<td>Globe</td>
<td>-0.7(±0.8)(^1)</td>
<td>-0.8(±0.8)(^2)</td>
<td></td>
<td>2.2(±0.8)(^3)</td>
</tr>
<tr>
<td>North</td>
<td>-2.1(±0.8)(^4)</td>
<td>-1.4(^5)</td>
<td>0.03(±0.5)(^6)</td>
<td></td>
</tr>
<tr>
<td>Tropics</td>
<td>1.5(±1.2)(^7)</td>
<td>-0.6(^8)</td>
<td>0.9 to 2.4(^8)</td>
<td></td>
</tr>
</tbody>
</table>

1. Plattner et al. 2002
2. −1.4 from Gurney et al. (2002) reduced by 0.6 to account for river transport (Sarmiento and Sundquist 1992; Tans et al., 1995)
3. Houghton, in press
4. −2.4 from Gurney et al. (2002) increased by 0.3 to account for river transport (Aumont et al., 2001)
5. −0.7 in forests (Goodale et al., 2002) and another equivalent amount assumed for non-forests (see text)
6. 1.2 from Gurney et al. (2002) increased by 0.3 to account for river transport (Aumont et al., 2001)
7. Undisturbed forests: −0.6 from Phillips et al. (1998) (challenged by Clark 2002)
8. 0.9 from DeFries et al., in review
1.3 from Achard et al. (2002) adjusted for soils and degradation (see text)
2.2(±0.8) from Houghton (in press)
2.4 from Fearnside (2000)

A longer-term estimate of the residual terrestrial C flux suggests that it was
nearly zero before 1935 (Fig. 1). That is, changes in land use explained most of the net
terrestrial C flux until about 1935. Since that time the residual terrestrial sink has
generally increased (Bruno and Joos, 1997).

This residual terrestrial sink may result from bias in the methods. For example,
limited quantitative understanding of the processes regulating oceanic O\(_2\) outgassing
may introduce biases in global sink estimates. It is difficult to simulate correctly the
contribution of the seasonal biosphere-atmosphere exchange to the observed
atmospheric CO\(_2\) field (‘rectifier effect’) (Denning et al., 1995) as required in
atmospheric transport inversions. The contribution of fluxes associated with the natural
C cycle to spatial gradients is debated (Taylor and Orr, 2000). Nevertheless, most
inverse analyses yield higher terrestrial C sinks than bottom-up, land-based approaches.
The resulting residual C sink has traditionally been attributed to environmental changes.

In contrast to the unknown bias of atmospheric methods, analyses based on
land-use change are deliberately biased. These analyses consider only the changes in
terrestrial C resulting directly from human activity (conversion and modification of terrestrial ecosystems). There may be other sources and sinks of C not related to land-use change (such as sinks caused by CO$_2$ fertilization) that exchange C with the atmosphere and are captured by other methods, but that are ignored in analyses of land-use change.

Figure 1. The annual change in global carbon storage in terrestrial ecosystems (Joos et al. (1999, updated), the annual flux from changes in land use (Houghton, in press), and the annual residual terrestrial sink (the difference between the changes in terrestrial storage and the land-use change flux). The values in the legend refer to the total change in C storage or C flux between 1850 and 2000. Negative values indicate a net terrestrial uptake of C. Changes in terrestrial C storage have been estimated by subtracting from fossil emissions the changes in the atmospheric C inventory, deduced from atmospheric and ice core CO$_2$ data, and in the oceanic C inventory, estimated with an ocean model.

A number of process-based terrestrial carbon models simulate the effects of CO$_2$, N, and climate on carbon storage (the effects commonly thought to account for the residual terrestrial flux). Examples include the four models described in a recent analysis by McGuire et al. (2001). Each of the models included some changes in land use as well as the effects of CO$_2$ and climate on terrestrial C storage. Although the models are very different from the bookkeeping model used by Houghton (in press), the analysis by McGuire et al. is nearly the complement of Houghton’s analysis: it includes environmental effects, while Houghton’s analysis does not; it does not consider changes in pastures, shifting cultivation, or logging, while Houghton’s analysis does (Table 3).
Thus, the two studies, together, appear to address both the flux of C from land-use change and the residual terrestrial flux. However, although process-based models generally find a global terrestrial C sink consistent with the magnitude of the residual terrestrial sink, the models are difficult to validate. Field observations of carbon sinks do not indicate the mechanism(s) responsible, and the relative importance of different growth-enhancing mechanisms varies among models. It is possible that much of the terrestrial sink attributed to environmental factors is, rather, the result of errors or omissions in analyses of land-use change, a possibility explored below.

Table 3. Terrestrial fluxes of C attributed to several mechanisms (average PgC yr\(^{-1}\) for the period 1980-1989)

<table>
<thead>
<tr>
<th>Mechanism</th>
<th>McGuire et al.*</th>
<th>Houghton</th>
</tr>
</thead>
<tbody>
<tr>
<td>Croplands</td>
<td>0.8**</td>
<td>1.21**</td>
</tr>
<tr>
<td>Pastures</td>
<td>NE</td>
<td>0.44</td>
</tr>
<tr>
<td>Shifting cultivation</td>
<td>NE</td>
<td>0.24</td>
</tr>
<tr>
<td>Logging</td>
<td>NE</td>
<td>0.29</td>
</tr>
<tr>
<td>Afforestation</td>
<td>NE</td>
<td>-0.10</td>
</tr>
<tr>
<td>Other***</td>
<td>NE</td>
<td>-0.11</td>
</tr>
<tr>
<td>CO(_2) fertilization</td>
<td>-1.9</td>
<td>NE</td>
</tr>
<tr>
<td>Climatic variation</td>
<td>0.4</td>
<td>NE</td>
</tr>
<tr>
<td>Total</td>
<td>-0.7</td>
<td>1.97</td>
</tr>
</tbody>
</table>

NE is ‘not estimated’. Negative values indicate a terrestrial sink.

* The estimates from McGuire et al. (2001) are the means of four process-based terrestrial carbon models.
** The simulations in McGuire et al. used net changes in cropland area from Ramankutty and Foley (1999) to calculate flux. The analysis by Houghton included gross rates of clearing and abandonment to calculate the flux attributable to croplands. The clearing of forests for croplands in the tropics exceeds the net increase in cropland area.
*** Fire suppression in the U.S. (-0.150 PgC yr\(^{-1}\)) and degradation of forests in China (0.044 PgC yr\(^{-1}\))

The only aspect of the studies (McGuire et al. and Houghton) that is redundant is the flux attributable to changes in the area of croplands. The difference in cropland fluxes (0.8 vs. 1.2 PgC yr\(^{-1}\)) (Table 3) is at least partly related to the manner in which the studies accounted for the expansion of croplands in the tropics (see the recent paper by House et al. (in press) for other differences). McGuire et al. used net changes in cropland area (from Ramankutty and Foley 1999) to determine annual rates of clearing. Houghton (in press) used rates of clearing based on changes in forest area reported by the FAO (2001). The net reduction in forest area often exceeded the increase in agricultural (cropland and pasture) area, and Houghton assumed that the excess deforestation resulted from simultaneous clearing (deforestation) and abandonment of croplands. That is, forests were cleared for new croplands, yet the area in croplands did not change correspondingly because croplands were also abandoned. The abandoned croplands were not reported as returning to forest (such lands often become degraded),
and thus the loss of forest area exceeded the increase in cropland. Because more forests were cleared for croplands under Houghton’s assumption than under McGuire’s assumption, the flux of C attributed to cropland expansion and abandonment was greater in Houghton’s analysis.

The difference in approaches points to the importance of accounting for all changes in land use. The area in croplands, and to a lesser extent, pastures and forests are often documented. Degraded lands are not. Thus, one can construct patterns of land-use change from agricultural statistics that miss important changes in C. Whether the sources and sinks are attributed to croplands or to degradation is of secondary importance. The important point is to capture the major changes in C (i.e., forests). Analyses based on satellite data rather than on agricultural statistics have the potential for full land-use accounting (Defries et al., 2002).

In summary, available bottom-up and top-down analyses, despite existing uncertainties, suggest a global net terrestrial sink, and a residual terrestrial sink. This conclusion is also supported by estimates of oceanic C uptake based on oceanic tracer data or models that are not explicitly discussed here (Prentice et al., 2001). Additional comparisons of terrestrial C fluxes can be obtained from a consideration of tropical and extra-tropical regions separately.

3.3 THE NORTHERN MID-LATITUDES

The net terrestrial C sink of ~0.7 PgC yr\(^{-1}\) for the 1990s is not evenly distributed over the land surface. Almost all analyses indicate that northern mid-latitude lands were a net sink, while tropical lands were a net source. As long as the north-south gradient in CO\(_2\) concentrations was the only constraint, the difference between the northern sink and the tropical source was defined, but the individual values were not (Tans et al. 1990). Thus, a sink of 2 PgC yr\(^{-1}\) in northern mid-latitudes and a source of 0 PgC yr\(^{-1}\) in the tropics could not be distinguished from a northern sink of 5 PgC yr\(^{-1}\) and a tropical source of 3 PgC yr\(^{-1}\). Based on an enlarged CO\(_2\) monitoring network, the sink in northern mid-latitudes for the 1990s is now thought to be 2.4 PgC yr\(^{-1}\) (Gurney et al. 2002) (not accounting for the riverine flux), which is offset to some degree by a net tropical source of 1.2 PgC yr\(^{-1}\). It has been estimated that rivers transport around 0.3 PgC yr\(^{-1}\) from the land to the ocean in the northern mid and high latitudes. Subtracting this riverine transport from the terrestrial net C-flux of 2.4 PgC yr\(^{-1}\) yields a northern terrestrial sink of 2.1 PgC yr\(^{-1}\).

One analysis of the distribution of atmospheric CO\(_2\) concentrations, without use of a transport model, suggests that much of the current terrestrial flux in northern extratropical regions is part of a natural circulation of C; and when the natural CO\(_2\) gradients are accounted for in transport inversion, the current (perturbation) sink is much smaller (<0.5 PgC yr\(^{-1}\)) (Taylor and Orr 2000) than suggested by others (Gurney et al., 2002) (see Conway and Tans (1999) for an alternative interpretation). A resolution of these estimates is beyond the scope of this analysis.

A recent synthesis of data from forest inventories found a net terrestrial sink of 0.7 PgC yr\(^{-1}\) for the northern mid-latitudes (Goodale et al., 2002). The estimate is less than half the sink for all northern mid-latitude lands inferred from atmospheric data, but if non-forest ecosystems throughout the region are as important in storing C as they seem to be in the U.S. (see below), this bottom-up analyses yields a sink (~1.4 PgC yr\(^{-1}\)) that is closer to the top-down estimate (2.1 PgC yr\(^{-1}\)). Pacala et al. (2001) reported a
similar overlap of top-down and bottom-up estimates for the U.S. Admittedly, the sink in non-forests is very uncertain. On the other hand, the northern sink of 2.1 PgC yr\(^{-1}\) from Gurney et al. is for 1992-1996 and would probably have been lower if averaged over the entire decade (see other estimates in Prentice et al. 2001). Top-down estimates for a short time period are sensitive to large interannual variations in the growth rate of atmospheric CO\(_2\). Another reason for potential discrepancies is that only C in trees is monitored in forest inventories, whereas changes in soil C may be equally or more important. On the other hand, the few field investigations that have considered soil C have found that soils account for only a small fraction (5-15\%) of the ecosystem’s C sink (Gaudinski et al., 2000; Barford et al., 2001).

Estimates of C-fluxes based on land use change statistics are around zero in the northern extratropical region (Houghton, in press). Taken at face value, this suggests that processes other than land-use change are responsible for a northern sink of 1 to 2 PgC yr\(^{-1}\). Next, we compare different bottom-up estimates available for the contiguous United States to investigate the plausibility of the estimate by Houghton.

### 3.4 THE UNITED STATES

Top-down estimates for relatively small regions such as the US are currently not very reliable. Gurney et al. (2002) estimate a C flux from the atmosphere to the land of 0.85 ± 0.5 Pg C yr\(^{-1}\) for the US. Based on an analysis of changes in land use, Houghton et al. (1999) estimated a C sink of 0.15 – 0.35 PgC yr\(^{-1}\) attributable to changes in land use. Pacala et al. (2000) revised the estimate upwards by including additional processes, but in so doing, they included sinks not necessarily resulting from land-use change. Their estimate for the uptake of C by forests, for example, was the uptake measured by forest inventories. If all of the accumulation of C in U.S. forests were the result of recovery from past land-use changes, then the uptake estimated from forest inventories should equal the flux estimated from land-use change statistics. The study by Caspersen et al. (2000) suggests that such an attribution is warranted. However, the analysis by Houghton et al. (1999) found that past changes in land use could account for only 10-30\% of the observed C accumulation in trees. The uptake calculated for forests recovering from agricultural abandonment, fire suppression, and earlier harvests was only 10-30\% of the uptake measured by forest inventories. The contributions reach 65\% if the uptake Houghton et al. attributed to woodland ‘thickening’ (0.52 PgC yr\(^{-1}\)) is included (Table 4). The results appear to be inconsistent with those of Caspersen et al. (2000).

The work of Caspersen et al. (2000) has been criticized by Joos et al. (2002) in two important respects. First, the relationship between forest age and wood volume (or biomass) is too variable to constrain the enhancement of growth to between 0.001\% and 0.01\% per year, as Caspersen et al. claimed. Enhancements of even 0.1\% per year yield estimates of biomass indistinguishable from those observed. Second, even a small enhancement of 0.1\% per year in net primary production would, for a doubling of CO\(_2\), yield a 25\% increase in growth (e.g., McGuire et al. (2001) in Table 3). Thus a small enhancement of growth may, nevertheless, translate into a significant C sink (Joos et al., 2002). Regrowth is clearly the dominant process in forests recovering from a disturbance, but C uptake by regrowth may be offset by C loss due to disturbances elsewhere (or at a later time). Hence, relatively small growth enhancement fluxes may play an important role for the net change in terrestrial C storage when considering large
HOUGHTON, JOOS AND ASNER

spatial areas and temporal scales longer than a few years. The question becomes: What is the current balance between C accumulation in regrowing forest versus the loss due to disturbances such as fire, storms, insects and how does growth enhancement affect this balance. Answering the question will be difficult, but one way to answer it is through the reconstruction of past human-induced and natural disturbances for the contiguous US and other regions of the globe.

Table 4. Estimated rates of C accumulation in the U.S. (PgC yr\(^{-1}\) in 1990)

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>low</td>
<td>high</td>
<td>low</td>
<td>high</td>
</tr>
<tr>
<td>Forest trees</td>
<td>-0.11</td>
<td>-0.15</td>
<td>-0.072***</td>
<td>-0.046****</td>
</tr>
<tr>
<td>Other forest organic matter</td>
<td>-0.03</td>
<td>-0.15</td>
<td>0.010</td>
<td>0.010</td>
</tr>
<tr>
<td>Cropland soils</td>
<td>-0.00</td>
<td>-0.04</td>
<td>-0.138</td>
<td>-0.00</td>
</tr>
<tr>
<td>Woody encroachment</td>
<td>-0.12</td>
<td>-0.13</td>
<td>-0.122</td>
<td>-0.061</td>
</tr>
<tr>
<td>Wood products</td>
<td>-0.03</td>
<td>-0.07</td>
<td>-0.027</td>
<td>-0.027</td>
</tr>
<tr>
<td>Sediments</td>
<td>-0.01</td>
<td>-0.04</td>
<td>NE</td>
<td>NE</td>
</tr>
<tr>
<td>Total sink</td>
<td>-0.30</td>
<td>-0.58</td>
<td>-0.35</td>
<td>-0.11</td>
</tr>
</tbody>
</table>

NE is ‘not estimated’. Negative values indicate a source of C to the atmosphere.

* Pacala et al. (2001) also included the import/export imbalance of food and wood products and river exports. As these would create corresponding sources outside the U.S., they are ignored here.
** Includes only the direct effects of human activity (i.e., land-use change and some management)
*** -0.020 PgC yr\(^{-1}\) in forests and -0.052 PgC yr\(^{-1}\) in the thickening of western pine woodlands as a result of early fire suppression.
**** -0.020 PgC yr\(^{-1}\) in forests and -0.026 PgC yr\(^{-1}\) in the thickening of western pine woodlands as a result of early fire suppression

Houghton et al. (1999) and Houghton (in press) may have underestimated the sink attributable to land-use change. Houghton did not consider forest management practices other than harvest (including regrowth) and fire suppression. Such activities as weed control, fertilization, breeding programs, and thinning have increased the productivity of US forests but were not accounted for. Neither did Houghton consider the proliferation of trees in suburban areas, or natural disturbances, which in boreal forests are more important than logging in determining the current age structure and, hence, rate of C accumulation in forests (Kurz and Apps, 1999). A fourth reason why the sink may have been underestimated is that Houghton used net changes in agricultural area to obtain rates of agricultural abandonment. In contrast, rates of clearing and abandonment are often simultaneous and thus create larger areas of regrowing forests than would be predicted from net changes in agricultural area. At present it is unclear how much of the C sink in U.S. lands can be attributed to changes in land use and management, and how much can be attributed to enhanced rates of growth.
One of the findings common to Houghton et al. (1999) and Pacala et al. (2001) is that non-forest ecosystems could account for a significant C sink. From 36-43% of the net sink estimated by Pacala et al. and about 74% of that estimated by Houghton et al. was in non-forests. Initially, Houghton et al. (1999) reported a sink of 0.14 PgC yr\(^{-1}\) in agricultural soils, an upper limit of 0.12 PgC yr\(^{-1}\) in woody vegetation expansion or encroachment and an upper limit of 0.05 PgC yr\(^{-1}\) in the thickening of woodlands. However, subsequent analyses of C accumulation resulting from conservation tillage in agricultural soils indicate a smaller sink (Schimel et al., 2000; Pacala et al., 2001). Furthermore, a more conservative estimate for woody encroachment and woodland thickening is half of the original upper limit proposed by Houghton et al. (1999). The reasons for the lower estimate are several fold. First, loss (rather than gain) of woody plants is occurring in some systems (Billings 1990). Second, a recent study of woody encroachment suggests that when changes in soil C are included, the displacement of grasses with woody shrubs may actually involve a net loss of C (Jackson et al., 2002), although this result remains highly contentious (Asner et al., in review). Somewhat independent of the belowground (soil organic C) responses to woody encroachment, the largest area of woody encroachment studied (> 40,000 ha in Texas) indicates an increase in aboveground C stocks of 0.02 Mg ha\(^{-1}\) y\(^{-1}\), accounting for both encroachment and management efforts to remove woody plants (Asner et al. in press). Extrapolated to the southwest U.S. region represented by this study (~ 50M ha), the results of Asner et al. suggest a net C sink from woody encroachment of about 0.001 PgC yr\(^{-1}\). Finally, a study of pine thickening in Colorado suggested that accumulation rates used by Houghton et al. may be too high (Hicke et al., in review).

Based on this and other emerging evidence, the ‘best estimate’ for the effects of land-use change (and fire management) on U.S. C storage is thus 0.11 PgC yr\(^{-1}\) (Houghton, in press). About 50% of this revised sink for the U.S. is attributable to changes outside of forests. Further, the sink in trees is only 40% of that estimated from forest inventories (Table 4). Thus, changes in land use yield a significantly lower sink than inferred from either inverse calculations or forest inventories. Either an enhancement in growth is equally important, Houghton’s analyses of land-use change have omitted some important management or disturbance processes, or the estimates from forest inventories and inverse calculations are too high.

This conclusion probably applies to all of the northern mid-latitudes. Both forest inventories and inverse calculations with atmospheric data show terrestrial ecosystems to be a significant C sink, while analyses of changes in land use show a net sink close to zero. The fraction of the northern C sink attributable to changes in land use and land management remains uncertain (Spiecker et al., 1996). It might be as high as 98% (Caspersen et al., 2000) or as low as 40% (Houghton, in press; Schimel et al., 2000). Resolution will require examination of forest age structure over more than two points in time and in regions other than those considered by Caspersen et al.; or it will require a more complete and spatially-detailed assessment of land-use change and land management in the U.S. and elsewhere. In any case, the impact of past human induced and natural disturbances on the evolution of terrestrial C storage needs to be investigated more carefully.
Do changes in tropical land use account for the net flux of C in that region? Inverse calculations show that tropical lands release on average a flux of 1.2 PgC yr\(^{-1}\) during the period 1992-1996 (Gurney et al., 2002). Accounting for a riverine transport of 0.3 PgC yr\(^{-1}\) (Aumont et al., 2001) yields a rate of decrease in terrestrial storage of 1.5 PgC yr\(^{-1}\). Because there are few air sampling stations over tropical lands, and because atmospheric transport over the tropics is not well understood, the error surrounding the flux estimate for the tropics is larger than it is for northern mid-latitudes.

A recent study by Townsend et al. (2002) combined atmospheric \(^{13}\)CO\(_2\) observations, modeling and land-cover change data to investigate pan-tropical C sources and sinks. Their analysis recognizes that deforestation in the tropics has led to an increase in the extent of C\(_4\) plants (warm climate grasses) relative to C\(_3\) plants (woody vegetation such as trees). Because C\(_3\) plants discriminate more strongly against \(^{13}\)C than do C\(_4\) plants, the latter leaves an atmospheric \(^{13}\)C signature more similar to that of the air-sea transfer of CO\(_2\) than to the land-air transfer (Rundel et al., 1989; Ciais et al., 1995). After accounting for the time-integrated replacement of \(^{13}\)C-rich organic matter in forest soils with \(^{13}\)C-poor organic matter in pasture soils (from C\(_4\) plant detritus), or “land-use disequilibrium”, Townsend et al. (2002) found that tropical regions were nearly C-neutral.

Forest inventories for large areas of the tropics are rare, although repeated measurements of permanent plots throughout the tropics suggest that undisturbed tropical forests are accumulating C, at least in the neo-tropics (Phillips et al., 1998). The number of such plots was too small in tropical African or Asian forests to demonstrate a change in C storage, but assuming the plots in the neo-tropics were representative of forests not disturbed by direct human interventions throughout the region yields a sink of 0.62 PgC yr\(^{-1}\). The finding of a net sink for the Amazon, however, has been challenged on the basis of systematic errors in measurement and plot size (Clark 2002; Keller et al., 2002). Phillips et al. (2002) counter that the errors are minor, but the results remain contentious. In sum, the two methods most powerful in constraining the northern net sink are weak or lacking in the tropics, and the C balance of the tropics is less certain.

Support for an accumulation of C in undisturbed tropical forests comes from some of the studies that have measured CO\(_2\) flux by eddy correlation (Grace et al., 1995; Malhi et al., 1998). Some studies suggest that the sinks in forests not disturbed by direct human interventions, if scaled up to the entire region, are larger than the emissions of C from deforestation (Malhi et al., 1998). Tropical lands would thus be a net C sink, but these results are controversial. The eddy correlation method for measuring CO\(_2\) flux includes both daytime and nighttime measurements. The direction of flux differs day and night and the micrometeorological conditions also differ systematically day and night. Wind speeds are much reduced at night, and CO\(_2\) efflux is negatively related to windspeed. If only those nights with the highest wind speeds are used to calculate an annual net flux, estimates change from a net sink to a net source of C (Miller et al., in press). Thus, large sinks in undisturbed forests are suspect. Some recent measurements of CO\(_2\) flux as well as measurements of biomass (forest inventory) do not show a large net sink (Rice et al., in press). In the Tapajós National Forest, Pará, Brazil, living trees were accumulating C, but the decay of downed wood released more C, resulting in a small net source from the site. The results suggest that
the stand is recovering from a disturbance several years earlier (Rice et al., in press; Keller et al., in review).

The net flux of C calculated from land-use change in the tropics is clearly a source of C to the atmosphere. Rates of deforestation are larger than rates of afforestation (FAO 2001). Based on data from the FAO, Houghton (in press) estimates that the net C flux resulting from deforestation, afforestation, and wood harvest in the tropics was a net source, averaging 2.2 PgC yr\(^{-1}\) during the 1990s. A sink of 0.43 PgC yr\(^{-1}\) was calculated for forests recovering from logging activities (Table 5), but this sink was more than offset by the large emissions from deforestation (and associated burning and decay of organic matter).

Comparing the results of different methods shows that the inverse calculations based on atmospheric data and models give a lower net C source from the tropics, just as they give a higher net sink in northern latitudes. In both regions, the sinks obtained through inverse calculations are larger than they are from analyses of land-use change. As discussed above, it is possible that much of the C released from land-use change is balanced by C sinks in forests not disturbed by direct human interventions. The sink (if it exists at all) might be the result of enhanced rates of growth, perhaps from CO\(_2\) fertilization, new inputs of nitrogen, or climatic variation. Alternatively, undisturbed forests are neutral with respect to C, and the source from deforestation is lower than Houghton estimates.

Table 5. Estimates of the associated sources (+) and sinks (-) of carbon (PgC yr\(^{-1}\) for the 1990s) from different types of land-use change and management (from Houghton, in press)

<table>
<thead>
<tr>
<th>Activity</th>
<th>Tropical regions</th>
<th>Temperate and boreal zones</th>
<th>Globe</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. Deforestation</td>
<td>2.110*</td>
<td>0.130</td>
<td>2.240</td>
</tr>
<tr>
<td>2. Afforestation</td>
<td>-0.100</td>
<td>-0.080</td>
<td>-0.180</td>
</tr>
<tr>
<td>3. Reforestation</td>
<td>0*</td>
<td>-0.060</td>
<td>-0.060</td>
</tr>
<tr>
<td>4. Harvest of wood</td>
<td>0.190</td>
<td>0.120</td>
<td>0.310</td>
</tr>
<tr>
<td>a. Wood products</td>
<td>0.200</td>
<td>0.390</td>
<td>0.590</td>
</tr>
<tr>
<td>b. Slash</td>
<td>0.420</td>
<td>0.420</td>
<td>0.840</td>
</tr>
<tr>
<td>c. Regrowth</td>
<td>-0.430</td>
<td>-0.690</td>
<td>-1.120</td>
</tr>
<tr>
<td>5. Fire suppression</td>
<td>0</td>
<td>-0.030</td>
<td>-0.030</td>
</tr>
<tr>
<td>6. Non-forests</td>
<td>0</td>
<td>-0.040</td>
<td>-0.040</td>
</tr>
<tr>
<td>a. Agricultural soils</td>
<td>0</td>
<td>0.020</td>
<td>0.020</td>
</tr>
<tr>
<td>b. Woody encroachment**</td>
<td>0</td>
<td>-0.060</td>
<td>-0.060</td>
</tr>
<tr>
<td>Total</td>
<td>2.200</td>
<td>0.040</td>
<td>2.240</td>
</tr>
</tbody>
</table>

* Only the net effect of shifting cultivation is included. The gross fluxes from repeated clearing of fallow lands and temporary abandonment are not included.
** Probably an underestimate. The estimate is for the U.S. only, and similar values may apply in South America, Australia, and elsewhere.
The existing data allow at least two, mutually exclusive explanations for the net tropical flux of C. One suggests that a large release of C from land-use change is partially offset by a large sink in undisturbed forests. The other suggests that the source from deforestation is smaller, and that the net flux from undisturbed forests is essentially zero. Under the first explanation, a growth enhancement (or past natural disturbance) is required to explain the large current sink in undisturbed tropical forests. Under the second, the entire net flux of C may be explained by changes in land use.

A third possibility, that the net tropical C source is larger than indicated by inverse calculations (uncertain in the tropics), is constrained by the magnitude of the net sink in northern mid-latitudes. As mentioned above, the latitudinal gradient in CO$_2$ concentrations constrains the difference between the northern sink and tropical source more than it constrains the absolute fluxes. Thus, the northern sink limits the magnitude of the net tropical source.

In summary, the evidence for a large C sink in the tropics (offsetting the source from land-use change) includes the uptake of C measured by eddy correlation techniques in undisturbed forests (Malhi et al., 1998) and the C accumulation observed on permanent plots in South America (Phillips et al., 1998, 2002). Evidence against a large sink in undisturbed forests includes biases in the measurements of CO$_2$ flux at many sites (Miller et al., in press; Rice et al., in press) and biases in measurement of change in biomass (Clark 2002). Next, we address potential biases in estimates of the land-use change flux.

3.6 POTENTIAL BIASES IN THE TROPICAL DEFORESTATION SOURCE

The high estimates of C emissions attributed to land-use change in the tropics (Fearnside 2000; Houghton, in press) may be too high. Potentially, there are at least three reasons: deforestation rates, tropical forest biomass, and rates of decay may each be overestimated. The rates of deforestation and afforestation used by Houghton (in press) to calculate a net flux are those reported by the FAO. The FAO uses expert opinion to determine the rates but must report a country’s official governmental estimate if one exists. It is somewhat surprising that the FAO would overestimate rates of deforestation. One can imagine that a country might want to underreport its rates of deforestation to appear environmentally ‘correct’. Why would it over-report the rate? Perhaps few countries insist on underreporting rates of deforestation, and the high estimates are, rather, the result of poor or biased data.

Two new studies of tropical deforestation, based on satellite data, report lower rates than the FAO and lower emissions of C than Houghton (in press). The study by Achard et al. (2002) found rates 23% lower than the FAO for the 1990s (Table 6). Their analysis used high resolution satellite data over a 6.5% sample of tropical humid forests, stratified by “deforestation hot-spot areas” defined by experts. In addition to observing $5.8 \times 10^6$ ha yr$^{-1}$ of outright deforestation in the tropical humid forests, Achard et al. also observed $2.3 \times 10^6$ ha yr$^{-1}$ of degradation. Their estimated C flux, including changes in the area of dry forests as well as humid ones, was 0.96 PgC yr$^{-1}$. The estimate is probably low because it did not include the losses of C from soils that often occur with cultivation or the losses of C from the degradation observed. Soils and land degradation (reductions of biomass within forests) accounted for 12 and 26%, respectively, of Houghton’s estimated flux for tropical Asia and America, and would
yield a total flux of $1.3 \text{ PgC yr}^{-1}$ if the same percentages were applied to the estimate by Archard et al. (2002).

The second estimate of tropical deforestation (DeFries et al., 2002) was based on coarse resolution satellite data (8km), calibrated with high resolution satellite data to identify % tree cover and to account for small clearings that would be missed with the coarse resolution data. The results yielded estimates of deforestation that were 54% less than those reported by the FAO (Table 6). According to DeFries et al., the estimated net flux of C for the 1990s was $0.9 \pm 0.4 \text{ PgC yr}^{-1}$.

If the tropical deforestation rates obtained by Archard et al. and DeFries et al. were similar, there could be little doubt that the FAO estimates are high. However, the estimates are as different from each other as they are from those of the FAO (Table 6). The greatest differences are in tropical Africa, where the percent tree cover mapped by DeFries et al. is most unreliable because of the large areas of savanna. On the other hand, the results may vary because the studies include different types of forests. Achard et al. considered only humid tropical forests; DeFries included all tropical forests. Both studies suggest that the FAO estimates of tropical deforestation are high, but the rates are still in question. The tropical emissions of C estimated by the two studies are about half of Houghton’s estimate: 1.3 and 0.9 \text{ PgC yr}^{-1}, as opposed to 2.2 \text{ PgC yr}^{-1}. Houghton’s estimate would be similar if based on these recent estimates of deforestation, lower than the FAO’s.

Fearnside’s (2000) and Houghton’s (in press) estimates of a tropical C source would also be high if their estimates of tropical forest biomass were too high. The biomass of tropical forests, particularly those forests that are being deforested or degraded, is poorly known (Houghton et al., 2000, 2001). Furthermore, logging, shifting cultivation, and other uses of forests are reducing the biomass of tropical forests and releasing C in the process (Brown et al., 1994; Flint and Richards, 1994). These processes of degradation may reduce the amount of C emitted through deforestation, but the loss of C is the same (with more coming from degradation rather than from deforestation).

Table 6. Percentage by which recent estimates of net change in forest area* for the 1990s are lower than reported by the FAO (2001)

<table>
<thead>
<tr>
<th>Region</th>
<th>Achard et al. (2002)</th>
<th>DeFries et al. (2002)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tropical America</td>
<td>18</td>
<td>28</td>
</tr>
<tr>
<td>Tropical Asia</td>
<td>20</td>
<td>16</td>
</tr>
<tr>
<td>Tropical Africa</td>
<td>42</td>
<td>93</td>
</tr>
<tr>
<td>All tropics</td>
<td>23</td>
<td>54</td>
</tr>
</tbody>
</table>

* The net change in forest area is not the rate of deforestation. It is the difference between the rate of deforestation and the rate of afforestation.

Finally, if downed trees take longer to decay and/or regrowth of biomass is faster than Houghton assumed, the calculated emissions from logging and deforestation may be overestimated (Monastersky 1999), especially in regions, such as Amazonia, where rates of logging have been increasing. In regions with a longer history of logging
and deforestation, using higher or lower rates of decay does not significantly change the calculated flux for the 1990s.

The major uncertainties in the tropics could be reduced with a systematic and spatial determination of rates of deforestation and afforestation, and with a systematic and spatial determination of biomass. In fact, a sensitive measure of biomass from space might help distinguish the changes in C attributable to land-use change from the changes attributable to enhanced growth --- at least for the aboveground biomass component. Measurement of changes in the amount of downed dead wood and soil C will require extensive ground measurements or a combination of ground samples and modeling (Chambers 2000; Keller et al., in press and in review).

4 Conclusions

There is a large body of evidence for a considerable global net terrestrial C sink. On the other hand, land use change analyses suggest a large global C source, implying an even larger (residual) C sink, more than offsetting the land-use source. In both the northern mid-latitudes and the tropics the terrestrial C sinks obtained through inverse calculations with atmospheric data are larger (or the sources smaller) than those obtained from bottom-up analyses (land-use change and forest inventories). Is there a bias in the atmospheric analyses? Or are there sinks not included in the bottom-up analyses?

For the northern mid-latitudes, when estimates of change in non-forests (poorly known) are added to the results of forest inventories, the result overlaps estimates determined from inverse calculations. Changes in land use include non-forest ecosystems but yield a smaller estimate of a sink. It is not clear whether management practices and natural disturbances, generally lacking in analyses of land-use change, are responsible for an additional sink, or whether environmentally enhanced rates of tree growth are responsible for the difference. Can the C sink in forests be explained by age structure alone (i.e., previous disturbances and management) (Caspersen et al., 2000), or are enhanced rates of C storage important (Houghton et al., 1999; Schimel et al., 2000)? In the tropics, the uncertainties are similar but also greater because atmospheric data for inverse calculations are more poorly distributed, and because forest inventories are lacking. Alternative approaches yield conflicting results concerning the enhancement of growth (or C storage) in undisturbed forests. Existing evidence suggests two possibilities. Either large emissions of C from land-use change are somewhat offset by large C sinks in undisturbed forests, or lower releases of C from land-use change explain the entire net terrestrial flux, with essentially no requirement for an additional sink. The first alternative (large sources and large sinks) is most consistent with the argument that factors other than land-use change or management are responsible for observed C sinks. The second alternative is most consistent with little or no enhanced growth. In both northern and tropical regions changes in land use exert a large influence on the flux of C. It is unclear whether other factors have been important. Of course, the relative importance of other factors need not be the same in both regions. The warmer temperatures in the tropics, for example, suggest that CO₂ fertilization would be more important there (Lloyd and Farquhar, 1996).

A resolution of the factors responsible for the residual terrestrial flux is of practical concern. If the sink is largely the result of management, it is more acceptable
as a C credit under the Kyoto Protocol, especially if it is the result of afforestation or reforestation. On the other hand, if the current sink is largely the result of forest regrowth, the sink is unlikely to persist for more than a few more decades (Hurt et al., 2002) without additional management practices to sequester C in, for example, degraded lands.

The magnitudes of the C sinks attributable to management, as opposed to environmental effects, could be evaluated in the northern latitudes with a spatial documentation of historical and current changes in land use, including the spatial extent of woody encroachment, and in the tropics with a systematic and spatial determination of rates of deforestation and afforestation, and, to a lesser extent, biomass. As much attention should focus on the fate of downed dead material as on the regrowth of secondary forests. The technical capacity for a systematic monitoring of forest cover in the tropics has existed for 30 years yet still needs to be implemented. The reasons go far beyond C accounting.

**Acknowledgements.** The authors thank Henry Gholz for helpful suggestions on an earlier version of the manuscript. Research was supported by NASA's Earth Science Enterprise's Land Cover/Land Use Change Program (NAG5-8637 and NAG5-11286), the NASA New Investigator Program (NAG5-8709), and the NASA LBA-Ecology Program (NCC5-481; LC-13). FJ thanks the Swiss National Science Foundation for support.

5 References


1 Introduction

Water quality and quantity are widely recognized as important environmental resources for society (Arnell et al., 2001). Quantity refers to the presence of a sufficient supply of fresh water to support the human and natural systems dependent on it, while quality refers to the suitability of the supply for its intended use (e.g. agricultural, domestic, industrial, or natural). Water is a dynamic substance, however, and the water cycle is a series of fluxes between reservoirs of varying size, residence time, and state. This interconnected property means that there are important consequences of the life history of water on a landscape, from its first appearance in precipitation to its exit to the ocean. The water carries a signature of its history (i.e., nutrient and pollutant loads) that has impacts and consequences on the reservoirs through which it passes.

Climate change will almost certainly impact the water cycle. These effects have been studied in some detail, although largely at global/regional scales; details are presented elsewhere in this book (Bonan et al., Chapter 17). For example, precipitation has increased 0.5-1% per decade in the 20th century in the mid-high latitudes of the northern hemisphere, with a greater frequency of heavy precipitation events, and these trends are likely to continue (IPCC Reports, 2001). This establishes the essential parameters of the water fluxes, but how that water is transported, allocated, and modified during residency on the land is an important research area for Land Cover and Land Use Change (LCLUC).

Land cover and land use are important determinates of the water supply on its transit through a landscape. Climate broadly establishes the upstream supply term of the water budget, and the effects of land cover and land use on water quality and quantity at the local level have been well established through numerous studies. For example, it is well documented that deforestation increases stream flow through decreased evapotranspiration (Bosch and Hewlett, 1982). Urban- and suburbanization also increase stream flow through increased runoff, but also decrease water quality when the amount of impervious surface in a watershed exceeds 10-15% of the total land cover (Schueler, 1994). The demand for water in all regions, but particularly in arid and semi-arid environments, for domestic, industrial, and agricultural uses has led to water engineering projects on all scales (Rosenberg et al., 2000). The net result is that demand often exceeds supply in these systems, resulting in some of the 20th century’s largest land transformations with consequent impacts on aquatic systems.
The case of the Aral Sea is one of the largest and well-known examples of the effects of water diversion (Micklin, 1994; Tanton and Heaven, 1999). Beginning in the 1960s, water from the two major rivers entering this inland sea (Amu Darya and Syr Darya) were diverted through engineering projects primarily for irrigation but also for storage, power generation, and flood control. This led to an enormous decrease in the area and volume of the Aral Sea over the next 35 years (Figure 1) from $6.7 \times 10^4$ km$^2$ to $3.2 \times 10^4$ km$^2$ and 1064 km$^3$ to 310 km$^3$ respectively (Saiko and Zonn, 2000). Most studies anticipate its complete disappearance in the next 25 years (e.g. Saiko and Zonn, 2000). Accompanying the shrinking sea is a long list of ecological and social impacts. For example, local climate has been impacted as evidenced by changes in surface temperature (Small et al., 2001a), while the exposure of saline soils and desertification...
has been accompanied by environmental health impacts (Small et al., 2001b). The enormous expansion in irrigated agricultural production, primarily in cotton, during the 1960s and 1970s provided an economic benefit from the water engineering projects. However, agricultural production has been declining over the last two decades due to salinization and water logging resulting from poor irrigation practices, and it has been estimated that nearly half of all irrigated lands \(8 \times 10^4 \text{ km}^2\) in the Aral Sea basin are affected (Heaven et al., 2002).

While case studies illuminate some of the direct cause and effect relationships between LCLUC and water quality and quantity, these relationships become more difficult to assess and quantify on larger scales (regional to national to continental to global). Global assessments of water supply and consumption are useful in cataloging the major categories of water demand and in pointing to the rise in consumption against a fixed supply (L’vovich et al., 1990). Yet the distribution of water on the Earth’s surface is not uniform and neither is the distribution of population or demand. Furthermore water distribution and demand do not necessarily co-vary. Therefore, from a land cover and land use perspective there is an unequal distribution of a resource for which the demand varies greatly (Vörösmarty et al., 2000). As the regions grow larger, the effects and ability to quantify become more diffuse, limiting the usefulness of aggregated global analysis.

Our focus in this chapter is to illuminate some of the relationships between LCLUC processes and water quality and quantity on the scale relevant to local and regional issues. Those relationships that affect climate/weather are treated by Bonan et al. (Chapter 17).

2 Water Quantity: Resource Allocation and Impacts

Water is a fundamental requirement of all living things, and thus water quantity has a direct impact on human and natural systems. Water vulnerability is typically cast as a human supply and demand problem, where climate and the landscape establish the supply and human systems make demands on the supply. Vulnerability is established when human demand exceeds some threshold in supply (Vörösmarty et al., 2000). However, natural systems are equally dependent on an adequate supply of water, and human systems have some dependence on natural systems for ecosystems goods and services (Costanza, 2001). In regions of the world where supply far exceeds demand, the main issues are on water engineering projects to control flooding or for power generation. The impacts from such projects primarily effect aquatic systems through dams that change the natural flow and channels of rivers (Rosenberg et al., 2000). In regions of the world where supply and demand volumes are more closely matched, the impacts are more diverse and can be more acute. This is a particularly persistent and growing problem in arid and semi-arid regions of the world, where terrestrial and aquatic biodiversity is concentrated along watercourses and in direct competition with the demands placed by the agricultural, industrial, and domestic sectors of society on the resource.

Given the scope of this topic, it is not possible to provide an exhaustive assessment of water quantity issues with respect to LCLUC. Rather, we will examine in some greater detail water allocation issues in arid and semi-arid regions to provide examples of the problems and relationships to LCLUC. Assessments of other processes
have recently been covered elsewhere (e.g. Rosenberg et al., 2000). The irrigation of cropland is the primary application of water in arid and semi-arid regions, and this has resulted in an enormous benefit to food security. For example, it has been estimated that expansion of irrigated lands increased food production by 50% between 1960 and 1985. By some estimates, 40% of current global food production is generated on the 15-20% of agricultural land that is irrigated (e.g. Shiklomanov, 1997). Thus irrigated lands are 3 times as productive as non-irrigated cropland, but also provide a greater value, as the dollar value of this production is 6-7 times that of non-irrigated cropland and >30 times that of rangeland (Crosson, 1997).

![Global Irrigated Land Area](image1)

![Global Irrigated Land Area](image2)

Figure 2. Global irrigated land area from 1800 to the present (top) and over the last 50 years (bottom). The rate of increase over the last 50 years is $3.4 \times 10^4 \text{ km}^2/\text{yr}$. Data compiled from Brown (1985, 1994), Brown et al. (1997), Warne (1970), Eckholm (1976), Postal (1994; 1999).

2.1 IRRIGATION AND LAND USE/LAND COVER

Global land area currently classified as irrigated agriculture has been estimated to be $2.5 \times 10^6 \text{ km}^2$. These are lands that were formerly covered by native ecosystems, used for grazing, or for dryland farming. As irrigated lands, they are now tilled, irrigated, and managed for agricultural production. The amount of irrigated land has increased
dramatically over the last two centuries (Figure 2) and has more than doubled between 1950 and 2000. It is not clear from the available data whether the rate of increase has slowed over the last decade, as there is some uncertainty on the estimates of total land irrigated and there is annual variability on the specific land parcels irrigated. Nevertheless the data shown in Figure 2b exhibit an apparent linear increase in the global estimates of irrigated land of $3.4 \times 10^4$ km$^2$ y$^{-1}$ over the last 50 years.

The specific parcels of land that are irrigated and managed for agriculture on an annual basis are not necessarily constant. Irrigated lands are abandoned due to myriad factors including loss of reliable water supply (e.g. diminished groundwater or reallocation of surface water) and degradation of the land (e.g. salinization and water logging). In the developed world the rate of abandonment is more or less balanced by the rate of development of new irrigated land resulting in a relatively stable total area of irrigated land over the last decade, while other parts of the world are experiencing explosive growth in irrigation, driven by huge government funded water engineering projects.

One area exhibiting expansion is along the Euphrates River in the Middle East (Syria, Turkey). Between 1990 and 2000, the amount of irrigated land increased 6 fold, from 175 km$^2$ to 1115 km$^2$, while the pattern of land use shifted where irrigated lands along the river bottoms and floodplains were abandoned for upland sites (Figure 3). The massive changes in water use, and attendant impacts on land cover, are being driven by a complex set of factors, including the governments’ desires to improve food security as well as to establish claims on the water resources (R. Smith, Yale University, pers. comm.).

2.2 IMPACTS OF RESOURCE ALLOCATION

The immediate environmental impacts of the reallocation of water resources (e.g. loss of wetlands, decline of aquatic species) have been well documented by a number of recent reviews (e.g. Lemly et al., 2000; Rosenberg et al., 2000; Vörösmarty and Sahagian, 2000). What is not as well understood or recognized are the impacts of shifting reallocations of water for irrigation or domestic/industrial use on native ecosystems. Particularly, what is the ability of native ecosystems to adapt and recover following abandonment of agricultural lands or through mitigation efforts to restore water to impacted systems? A series of studies by Elmore et al. (2003a; 2003b) provide important insights into this question.

Elmore et al. (2003a; 2003b) studied the behavior of semi-arid ecosystems in the Owens Valley of California. Owens Valley has a 130-year history of land use centered on the abundant fresh water resources in this semi-arid landscape. Agricultural activity (irrigated crops and pasture) peaked in the 1920s followed by large-scale abandonment due to a reallocation of the water resources, through interbasin transfer, for domestic/agricultural/industrial use by Los Angeles, CA. This water makes up a significant fraction of the fresh water budget for Los Angeles, and virtually all of the surface runoff has been exported from the valley since the 1920s. Following abandonment, much of the agricultural land was colonized by a mixture of perennial shrubs and annual grasses and plants.
The dramatic decrease in surface run-off during the drought that lasted from 1986 through the early 1990s prompted a response from resource managers to increase the amount of water extracted from groundwater reserves. At the height of the drought groundwater made up a significant fraction of the total water exported from Owens Valley. Elmore et al. (2003a; 2003b) examined the response of the semi-arid ecosystems to these two forcing functions (drought, groundwater extraction) using a combination of remotely sensed and field data. They found that, not surprisingly, vegetation communities not dependent on groundwater (xeric) as well as those dependent on groundwater (phreatophytic) but where the groundwater levels remained within 3 meters of the surface were little affected by the drought. However, where groundwater dropped below 3.3 meters due to groundwater withdraw, phreatophytic meadow communities displayed catastrophic decreases in plant cover, indicating that a threshold had been reached.

Following the return of average to above average precipitation during the mid to late 1990s, the communities most affected by the groundwater draw down exhibited changes in plant species assemblages, with a shift towards invasive, non-native annuals and opportunistic shrubs. Furthermore, it could be clearly demonstrated that prior land
land use (agriculture) had a legacy that was evident 80 years after land abandonment. Previously cultivated lands showed a higher abundance of opportunistic shrubs and non-native annual plants, and lower species diversity than lands that were never cultivated. The vegetation on these lands also showed an amplified response in green activity to annual rainfall compared to the land dominated by native species. This response was detected with remotely sensed data where large increases in the amount of green cover during wet years were followed by large decreases in green cover during dry years. Thus the drought and the groundwater pumping changed the fundamental ecological functioning characteristics of these areas. It is unknown if the observed impacts represented a permanent shift as these studies only cover a twenty year period. However, ecological impacts of prior agricultural activity on lands abandoned for 80 years, as well as impacts from groundwater withdrawn in the 1970s, strongly suggest that these shifts are long-term if not permanent.

The impacts resulting from the abandonment of irrigated agricultural land have also been documented in regions where groundwater is far from the surface (Okin et al., 2001). In addition to changes in species composition similar to that documented in Owens Valley, there are significant impacts on soil properties, which might explain the persistence of the land-use legacy in Owens Valley. Soils destabilized by agriculture become susceptible to wind erosion following abandonment causing deflation in the former fields and deposition downwind. This results in changes in the natural processes of nutrient accumulation and availability through deflation and burial aggravating the impacts. Thus there are direct and indirect impacts of abandonment that persist over the long-term.

These example case studies provide detailed insight into the immediate impacts of water resource allocation and the legacy of land use. Other analyses not reported on here support these basic principles, but also extend the reach of our understanding into regions with diverse socio-economic drivers and land use histories. While the specific details in these region differ, the common threads are that in arid and semi-arid regions water vulnerability is high, agricultural abandonment is relatively common due to degradation and/or shifting priorities in allocation, and the margins for resilience (capacity to withstand climatic variability) are shrinking in the presence of high water demand.

2.3 REGIONAL TO GLOBAL IMPLICATIONS

The case studies described above provide concrete examples of the relationship between LCLUC and water resources. Recognizing that these examples are not a complete assessment of all possible drivers and impacts related to LCLUC and water quantity, it is apparent that a basic framework for assessing these problems is understood. In humid and temperate regions where upland ecosystems as well as agriculture are sustained by rainfall, LCLUC impacts on hydrology are focused around drainage systems. Here direct impacts are on the riparian and aquatic systems through water engineering (Rosenberg et al., 2000). In regions where ecosystems are water limited and agriculture sustains a substantial benefit by irrigation, LCLUC impacts can extend many tens of kilometers from the actual water source and affect large land areas, in addition to the direct impacts on riparian and aquatic systems.
While the impacts of LCLUC on water quantity are directly measured and understood at the local scale, assessments of water vulnerability tend to be made at the regional, national, or global level (e.g., L’vovich and White, 1990). Such assessments are typically cast as coarse supply and demand problems, the analysis of which can be conveyed conveniently in simple graph or tabular formats. However, advances in technology and analytical capabilities are opening up new avenues to assess water vulnerability. Vörösmarty et al. (2000) presented the first geographically explicit global assessment of water vulnerability. Using a water balance model validated to a 1985 database of river discharge, they compute runoff at a 30’ (latitude by longitude) resolution. A spatially explicit population distribution was derived from several sources and matched to the discharge database. The last step was to then match the population/runoff calculations with spatially explicit water demand calculations derived from county-level water withdrawal statistics. They were thus able to produce a global map of population and water demand and apply a stress threshold (demand approaching or exceeding supply) to show a global distribution of water stress as a function of population (Figure 4). The resultant map shows regions of the world where water quantity is stressed due to human demands, and where LCLUC related to water quantity issues are likely to be pervasive. This map clearly highlights vulnerable regions in western North America, areas bordering the Sahara Desert, the Arabian Peninsula, and the several densely populated areas in Asia (Pakistan, India, and northeast China). These are not surprisingly also contemporary regions of important LCLUC impacts associated with water quantity.
3 Water Quality

3.1 SOME BASIC HYDROLOGY AND BIOGEOCHEMISTRY

Stream flows typically have two main components. There is a base flow resulting from previous rain water infiltrated into soils and groundwater as well as storm flow resulting from direct rainfall inputs to the stream surface and from overland flow during rain events. Base flow is the slow, but continuous discharge generated by groundwater inputs to streams between storm events, and storm flow is a rapidly changing component of discharge which creates hydrographs responding to rain events (Figure 5).

![Storm hydrograph Feb. 1984, Choptank River](image)

Figure 5. Example of a storm hydrograph observed at USGS gauging station 01491000 at Greensboro MD from a rain event centered on 15 Feb 1984 (water year day 137). Data from Fisher et al. (1998).

Water yields are the quantity of stream flow per unit watershed area per unit time. Units for water yields are all equivalent to depth time$^{-1}$, which is equivalent to volume discharged per unit time per unit watershed area (cm$^3$ water cm$^{-2}$ watershed area y$^{-1}$ or cm y$^{-1}$). Water yields may also be expressed as ft$^3$ s$^{-1}$ mile$^{-2}$ or inches y$^{-1}$ (reported in USGS Water Resources Reports). Water yields reported as depth time$^{-1}$ (cm y$^{-1}$ or inches y$^{-1}$) are immediately comparable to rain inputs (also in cm y$^{-1}$ or inches y$^{-1}$) and lead to a simple evaluation of the fraction of rainfall that is discharged from a watershed as surface water discharge. In mesic areas with precipitation of $\sim$1 m y$^{-1}$ (e.g., much of the US east of the Mississippi River), water yields are $\sim$40 cm y$^{-1}$ or about 40% of rainfall. This ratio varies from 0.3 to 0.6, increasing with latitude as mean annual temperature decreases (Van Breeman et al. 2002). The remainder of the rain water not exiting a basin as stream flow is returned to the atmosphere by (1) evaporation (a physical process), (2) plant transpiration (a biological process involving water vapor loss during photosynthesis), or (3) deep seepage losses from the watershed.
to regional groundwater (a geological process, usually small compared to evaporation and transpiration). Evaporation + transpiration, collectively referred to as “evapotranspiration” or “ET”, are often lumped as one vapor loss term with the same units as water yield.

Elemental export from a watershed is estimated as stream flow x concentrations. Typically, continuous or near-continuous stream flow data are available, with a limited number of observed concentrations. Flow-weighting or other estimators are used to combine these data to calculate export at a time scale of months or years (e.g., export = flow (m$^3$ y$^{-1}$) * average concentration (g m$^{-3}$) = g y$^{-1}$). As for discharge, export may also be normalized to basin area. Termed “export coefficients”, the SI units are usually kg ha$^{-1}$ y$^{-1}$ or kg km$^{-2}$ y$^{-1}$; also used are lbs acre$^{-1}$ y$^{-1}$ (~equivalent to kg ha$^{-1}$ y$^{-1}$). Export coefficients are essentially areal fluxes of materials exported from a watershed, and can easily be compared with watershed inputs such as fertilizer application rates (e.g., lbs acre$^{-1}$ y$^{-1}$ or kg ha$^{-1}$ y$^{-1}$). Thus, as for water yields, elemental or other export coefficients can also be expressed as a fraction of inputs. Export coefficients are also useful as a modeling tool because they are usually strongly influenced by land cover and land use, as described below.

3.2 EFFECTS OF LAND USE ON WATER YIELDS

Water yields are strongly influenced by land use and land cover. Due to higher plant biomass, forests usually have much higher rates of evapotranspiration than agricultural or urban land uses, leaving less water available for groundwater or overland flows to streams. As a result, conversion of forest land cover to anthropogenic land uses in mesic areas of the temperate zone usually results in increased water yields (Figure 6). These observations are based on forest logging in mesic to wet areas (Bosch and Hewlett, 1982), and there was no significant difference between removal of conifer and hardwood forests. Water yields increased by an average of 20 cm y$^{-1}$ in watersheds with complete forest removal. Considering that watersheds with mixed land cover currently have water yields of ~40 cm y$^{-1}$ (e.g., Van Breemen et al., 2002), removal of forest land cover clearly has significant impacts on water yields.

Urban land cover also influences water yields. Impervious surfaces such as roads, roofs, and parking lots cause storm responses to be faster, with higher discharges and power for bank erosion. Impervious surfaces directly shunt rainfall to overland flow and prevent infiltration to groundwater. The decreased groundwater inflows cause a rapid drop in flows after the end of the rain event, and the reduced abundance of plants results in much lower evapotranspiration. The net effect is that storm hydrographs have a larger volume of water over a shorter period of time in urbanized areas, with reduced base flows between storm events.

Forested land cover is the most retentive of water as well as particulates and dissolved materials. It is clear from the above examples that forested landscapes produce the most steady and smallest stream discharges of any land cover. Conversion of forest to anthropogenic land uses increases both the total flow (Figure 6) as well as the volume and erosive power of storm flows while decreasing base flows. Similarly, forested landscapes also export the smallest amount of dissolved and particulate material, whereas anthropogenic land uses such as agriculture and urban areas export much higher amounts, as described below.
Figure 6. Water yields from temperate watersheds with varying amounts of conversion of forest to anthropogenic land uses. Water yields increase with less forest cover due to lower ET. Data source: Bosch and Hewlett (1982).

3.3 EFFECTS OF AGRICULTURE ON NUTRIENT YIELDS

The agricultural production of food for human populations is an essential need of modern societies. Within the last 100 years, two major changes in agriculture have occurred in North America and Europe: (1) since the beginning of the 20th century a largely agrarian population has gradually shifted to an urban base, with a large fraction of the population in urban areas dependent on a small number of agricultural workers; and (2) after World War II there were large increases in crop yields from smaller areas of land using commercial fertilizers containing primarily nitrogen (N), phosphorus (P), and potassium (K). These changes have concentrated the production of food into smaller areas than in the 19th century, reduced the number of people involved in food production, and greatly increased the intensity of agricultural land use.

One of the unintended consequences of the increased intensity of agricultural land use has been the contamination of shallow groundwater with nitrate (Figure 7). Inexpensive commercial fertilizers became available after World War II as munitions factories were converted to the production of agricultural N fertilizers, and application rates on agricultural lands increased exponentially for 30 years (Figure 7, lower panel). Sampling and dating of groundwater on the agriculturally dominated Delmarva Peninsula by Bohlke and Denver (1995) showed historical increases in groundwater nitrate that largely paralleled the fertilizer N application rates (Figure 7, upper panel). Nitrate concentrations of 10-20 mg NO$_3$-N L$^{-1}$ are frequently observed in most shallow aquifers in North America in agricultural areas (http://waterdata.usgs.gov/nwis/gw),

\[ r^2 = 0.21 \]
and these waters are undrinkable due to the tendency of nitrate concentrations >10 mg NO$_3$-N L$^{-1}$ to cause methemoglobinemia in infants and to form carcinogenic nitrosamines in the human intestine (US EPA 1976, Heathwaite et al., 1993).

Figure 7. Historical record of fertilizer sales and groundwater nitrate under agricultural areas on the Delmarva Peninsula. Fertilizer sales in each county are assumed to represent application rates.

Nitrate (NO$_3$) is almost never applied as N fertilizer. Urea or ammonia (NH$_3$) are more common, but bacterial processes in soils oxidize urea and NH$_3$ to produce highly soluble NO$_3$. This process, termed “nitrification”, is a natural heterotrophic response to excess N, which yields energy for the nitrifying organisms (e.g., Aber 1992). The result is that readily sorbed forms of N (urea, NH$_3$) are converted to highly soluble NO$_3$ by biological activities in soils. During late fall and early winter when hydrologic connections are reestablished after the seasonal lowering of groundwater in summer, rain water infiltrating to groundwater may carry high concentrations of nitrate (1.5-15 mM NO$_3$ or 20-200 mg NO$_3$-N L$^{-1}$) from the root zone of heavily fertilized agricultural areas into shallow aquifers (Staver and Brinsfield, 1998). This transfer of nitrate from the root zone to groundwater is a flushing of excess N, which can be greatly reduced by winter cover crops that maintain some plant N uptake during the cold season (Staver and Brinsfield, 1998).
Over the last twenty years, fertilizer applications in North America appear to have largely stabilized (e.g., Figure 7). Furthermore, in the late 1990's concerns about nitrate in groundwater have prompted the US Dept. of Agriculture (USDA) to require nutrient management plans to reduce groundwater nitrate in agricultural areas, and the Conservation Reserve Enhancement Program (CREP) now provides funds to create riparian zones along streams to intercept nitrate in contaminated groundwater. However, the multi-year residence time of most groundwater (Focazio et al. 1998) means that a decade will probably be required for the effects of nutrient management to be readily observable.

Other land uses also influence groundwater nitrate (Figure 8). Although agricultural land use has the highest nitrate concentrations in groundwater, unsewered urban areas have similarly high NO$_3^-$, whereas forested areas have the lowest amounts of NO$_3^-$ due to lower inputs and uptake by plants. The production of human food (agriculture) and the disposal of human waste in septic systems are clearly the principal causes of elevated NO$_3^-$ in groundwater.

Due to the large differences in groundwater NO$_3^-$ under varying land uses (Figures 7-8), the NO$_3^-$ concentrations of streams reflects the spatially varying distributions of land uses in watersheds (Figure 9). These stream concentrations from the Choptank River basin on the Delmarva Peninsula are annual means of TN; NO$_3^-$ is the single largest component of the TN, represents ~70% in streams on the Delmarva Peninsula (Fisher et al., 1998; Norton and Fisher, 2000). Note that the effect of decreasing forest cover and increasing agriculture is not linear, particularly at high % agriculture and low forest. As agriculture expands to >70% of land cover, normal landscape traps for NO$_3^-$ (e.g., wetlands, forested areas along streams) are converted to agriculture, and NO$_3^-$ concentrations in streams rise exponentially. The effect of agriculture on stream chemistry is not limited to the agriculturally dominated Delmarva Peninsula; in a watershed survey of the mid-Atlantic region, Jordan et al. (1997) showed relationships in coastal plain streams similar to those in Figure 9, with similar concentrations, although the data of Jordan et al. (1997) are <70 % agriculture and appear linear. Better drained piedmont streams have even higher concentrations of NO$_3^-$ for the same amount of agriculture (Jordan et al., 1997).

Despite the strong relationships described above, not all NO$_3^-$ in groundwater appears in baseflow. Using the average land use specific NO$_3^-$ concentrations in Figure 9, Lee et al. (2000) have estimated the land use adjusted NO$_3^-$ in surface groundwater of sub-basins of the Choptank watershed and compared these to base flow nitrate of streams draining these sub-basins. In 34 sub-basins, baseflow nitrate was less than the estimated groundwater nitrate, and the fractional decrease in observed baseflow nitrate
was a linear function of the amount of hydric soils in each sub-basin. These water-saturated soils have little air space, are often anoxic, and are probably sites of denitrification (conversion of NO$_3^-$ to N$_2$ gas) by soil bacteria using NO$_3^-$ as an alternate electron acceptor in the absence of O$_2$. Thus, there appears to be some natural attenuation of anthropogenic NO$_3^-$ by hydric soils in watersheds, and NO$_3^-$ export may be reduced by as much as 80% below the expected concentrations based on land use (Lee et al., 2000).

Figure 9. The effect of varying land use (agriculture and forest) on stream nitrate concentrations in subbasins of the Choptank River watershed. Data of Norton and Fisher (2000).

Agricultural land use also influences the other two major components of commercial fertilizer. Potassium (K$^+$) is also highly soluble, and Driscoll and Whitall (unpub.) have reported a strong positive correlation between potassium concentrations in streams and % agricultural land use in New England basins (D. Whitall, pers. com.). Phosphorus, however, unlike the highly soluble NO$_3^-$ and K$^+$, accumulates in surface soils due to strong sorption reactions of H$_2$PO$_4^-$ with positively charged iron oxyhydroxide and aluminum oxide in soils, particularly when excess animal manures are applied (Sims and Wolf 1994). Although the sorbed P is not transported vertically downward to groundwater, erosion of the highly enriched surface layer of agricultural soils results in considerable horizontal P transport during storm events. NO$_3^-$ concentrations are often reduced during the storm flow because overland flows contain less NO$_3^-$ than the groundwater which contributes baseflow; however, the overland flows transport eroded soils with high P concentrations, increasing stream P dramatically during storms (e.g., Fisher et al., 1998). Thus N, P, and K are transported from watersheds in very different modes; N and K are continuously transported primarily in baseflows, whereas P transport is more episodic and related to storm events.

A review of studies of relatively small areas of nearly uniform land use (Beaulac and Reckhow 1982) has provided a synthesis of land use specific N and P yields. As for water yields, forests have the lowest export coefficients of N and P by any land cover, and anthropogenic land uses along an approximate gradient of increasing disturbance show increasing export coefficients for N and P over several
orders of magnitude. Small, but severely disturbed areas such as animal feedlots have export coefficients ~100 X higher than other types of agricultural land use.

3.4 EFFECTS OF HUMAN POPULATIONS ON NUTRIENT EXPORT

Human populations also directly influence nutrient concentrations and export from basins. As shown above in Figure 8, septic systems in less densely populated areas oxidize organic N from human waste and produce NO\textsubscript{3}, enriching local groundwater. Valiela et al. (1992) have shown that increasing housing density in unsewered areas on the sandy soils of Cape Cod results in increasing NO\textsubscript{3} in groundwater. In more densely populated areas served by sewage systems, N may be discharged in many forms, usually to surface waters (e.g., Ryding and Rast, 1989; Fisher et al., 1998), and these discharges may represent a large fraction of the N and P inputs to aquatic systems (e.g., Lee et al., 2001). Efforts to reduce eutrophication of aquatic systems usually begin with diversion or upgrading of treatments in sewage systems to reduce N and P inputs (e.g., Ryding and Rast, 1989). In large river basins, the integrated impacts of human populations (both from agricultural production of food and disposal of human wastes) result in strong positive correlations between human population density and river NO\textsubscript{3} concentrations and export (Peierls et al., 1991).

3.5 MODELING OF WATERSHED HYDROLOGY AND CHEMISTRY

The empirical studies described above provide a strong conceptual basis for hydrochemical modeling of watershed export. While it can be argued that models tell us only what we already know, it is also true that models can be used to conduct “experiments” (model scenarios) that are empirically impractical, but which incorporate empirically derived principles such as those summarized above. For example, a hydrochemical model which has been calibrated under current conditions may be used to simulate the effects of manipulations which could never be done in practice within a watershed; e.g., removal of human populations, changes in human waste disposal, conversion of all land uses to forest, urbanization, etc. While the results of such model scenarios are predictions based on inputs often well outside of the calibration conditions, the projected conditions may be useful to select the best of several possible options, as in EPA’s current TMDL process.

We have used the hydrochemical model Generalized Watershed Loading Functions (GWLF) to examine the effects of land use/cover change in the Choptank River basin. The model was carefully calibrated at a USGS gauging station over an 11 water year period with detailed stream chemistry data using many of the empirical principles described above (Lee et al. 2000), and at the annual time scale the model has estimated validation errors of 5-10% for export of water and N and ~35% for export of P (Lee et al. 2001). We have used this model to estimate the biogeochemical effects of land use changes in the Choptank basin over the last 150 years observed in historical maps, aerial photographs, satellite imagery, and human population data from the US Bureau of the Census (Benitez 2002, Benitez and Fisher in revision). Using the estimated land cover and population changes, we have performed model manipulations to isolate the effects of fertilizer applications, human waste, and urbanization on watershed nutrient export. Below we describe the model results from a sub-basin within
the Choptank, which has experienced fertilizer applications as well as urbanization and growth of human populations.

Tilghman Island is an island of 5.9 km² connected by a bridge to a peninsula of land which lies between the main axis of Chesapeake Bay and the Choptank estuary, a tributary of Chesapeake Bay. In 1850 Tilghman Island was primarily forest and agricultural lands, with small amounts of wetlands and urban areas, and a human population density ~0.05 ha⁻¹ (Figure 10). Between 1850 and 1980 there was a net loss of forest and increases in agriculture and urban areas as the population density increased to ~0.5 ha⁻¹. During this period, application of fertilizers to agricultural lands increased by several orders of magnitude at a rate similar to other regions of the US (Goolsby et al., 1999). However, beginning in ~1980, the prime location of Tilghman Island on the Chesapeake encouraged the development of tourism, and urbanization and human population densities increased to ~30% and >1 ha⁻¹, respectively, primarily at the expense of agricultural land. Thus, this small region of the Choptank Basin has experienced both the conversion of the original forest cover to agriculture as well as significant urbanization.

Figure 10. Historical changes in land use, human populations, and fertilizer application rates at Tilghman Island estimated from historical maps, aerial photographs, satellite imagery, and census data (Benitez 2002; Benitez and Fisher, in review).

We used the GWLF model to estimate the effects of these changes on export of water, N, and P from this basin. We used the observed land use/cover changes (Figure 10) as inputs to GWLF to estimate the overall effect of the land use changes, and then we used model scenarios in which we manipulated the history of Tilghman Island for comparison with the full model scenario to separate the individual effects of
conversion to agriculture, application of fertilizers, and disposal of human wastes (Figure 11).

The first model scenario was the estimation of N and P export under forest prior to European colonization. Using the advantage of modeling, we removed the people, agriculture, and urban areas, and reforested the entire basin. The estimated export coefficients under these conditions were quite low, 0.7 kg N and 0.09 kg P ha\(^{-1}\) y\(^{-1}\), approximately 20 and 4 times lower than those currently estimated (18 kg N and 0.4 kg P ha\(^{-1}\) y\(^{-1}\)). Although this model scenario is farthest from the calibration conditions, these estimated increases in nutrient export indicate the magnitude of the anthropogenic effects on Tilghman Island and are consistent with currently observed exports of forests (Clarke et al., 2000). Other, less manipulative model scenarios were used to estimate the individual effects of conversion of forest to agriculture, application of fertilizers, and disposal of human wastes.

Figure 11. Changes in N and P export from Tilghman Island using the hydrochemical model GWLF.

Two effects of agriculture were examined. Prior to 1850, conversion of forest to agriculture was not accompanied by application of fertilizers, although manures, guano, and lime were increasingly employed later in the 19\(^{th}\) century (Benitez and Fisher in review). Thus, the model output for 1850 conditions under low population density and ~60% agriculture is an estimate of the effect of the conversion of forest to a primarily agricultural landscape. Both N and P export appear to have increased ~2 fold from the all-forest scenario (green arrows in Figure 11) to ~1.2 kg N and 0.23 kg P ha\(^{-1}\) y\(^{-1}\). The subsequent application of fertilizers to agricultural lands in the second half of...
the 20th century had a much greater impact on N export yields than the initial conversion to agriculture, although this was not true for P export. Comparing the full model scenario with the one in which fertilizers were withheld, the model results suggest that application of agricultural N fertilizer was responsible for 30-70% of the historical increases in N yields. P export was little affected by withholding of fertilizers because storm flows do not erode large amounts of surface soils under the relatively low relief of Tilghman Island (maximum elevation 3 m asl). In areas with greater topographic relief, a larger effect of storm flows would be expected. Thus conversion of forest to agriculture in the 18th and 19th centuries appears to have increased N and P export coefficients by a factor of 2, but application of fertilizers in the 20th century resulted in a factor of ~5 increase in N export and an almost undetectable effect on P export.

The effect of waste disposal from human populations was estimated in another model scenario. Human populations were permitted to develop as observed in census data, but all waste discharges were set to zero. For simplicity in Figure 11, we show the combined effects of withheld fertilizer applications and human waste (lowest lines in each panel). For both N and P, increases in export yields were small compared to the 1850 conditions (<2 kg N and ~0.3 kg P ha\(^{-1}\) y\(^{-1}\)), indicating that most of the estimated increases in nutrient yields from Tilghman Island for the 1990's was due to application of fertilizers to agricultural lands and disposal of human wastes. The effect of the latter was estimated to be about a doubling of N export yields and an increase of ~30 % for P export yields.

4 Summary

The specific drivers and effects of land use and land cover change on water quantity and quality tend to be specific to each region. For example, water diverted from a river for domestic use may be returned to the same river through municipal effluent, used for irrigation and thus largely evaporated, or removed entirely from the system by interbasin transfer. Nevertheless, as we have outlined this chapter, there are commonalities relevant to the impacts of LCLUC on hydrology: land cover directly affects stream and river flow, and land use directly impacts water quality. In particular, we have identified the production of food in agricultural areas and the disposal of human waste from populated areas as the primary causes of degraded water quality. As a result, many regions of the world are reaching a state of vulnerability with respect to water quantity and quality, and this situation is likely to worsen over the next 25 years, particularly if human populations continue to increase.
5 References


CHAPTER 16

LAND USE CHANGE AND BIODIVERSITY
A Synthesis of Rates and Consequences during the Period of Satellite Imagery

ANDREW J. HANSEN1, RUTH S. DEFRIES2, WOODY TURNER3

1 Montana State University, Ecology Department, Bozeman, MT 59717
2 University of Maryland, Earth Systems Science Interdisciplinary Center, Computer and Space Sciences Building and Department of Geography, 2181 Lefrak Hall, College Park, MD 20742
3 NASA Office of Earth Science, Mail Code YS, Washington, DC 20546

1 Introduction

The expansion and intensification of human land use in recent decades is resulting in major changes in biodiversity. Biodiversity, a term that has entered into common usage only in the last twenty years, refers to the diversity of life at all levels of organization, from genetic to species to ecosystem (Levin 2000). Although we refer throughout this chapter most commonly to species diversity, land use change alters biodiversity at all of these levels. For example, reduced habitat from land use change decreases population sizes and reduces genetic diversity within a species. At the other extreme, land use change commonly leads to more homogenous landscapes reducing ecosystem diversity (Flather et al., 1998). The advent of remotely sensed data from satellites has provided a basis for quantifying rates of land use change around the world and consequences on biodiversity. Global satellite-based data sets, useful for analyses of rates of land use change since the 1970s, are now becoming available. These analyses reveal that in these recent decades, land use continues to intensify in formerly occupied areas and expand into what were formerly natural habitats. This paper aims to summarize what we have learned about interactions between land use and biodiversity, especially during the period of satellite data availability.

Mustard et al. (this volume) synthesizes four phases of land use along a trajectory of economic development. These are: natural, frontier (resource extraction from natural ecosystems), agricultural expansion, and industrialization and urbanization. These phases are associated with a trajectory of land use change, which generally proceeds from wildlands, through increasingly intense land use to urban uses (Figure 1). While many temperate and tropical landscapes have been heavily affected by land use for centuries, rapid change among each of these four phases of economic development and the resulting land use trajectories continues today.

Many tropical forests are in the frontier stage, undergoing initial logging and conversion of wildlands to extractive plantations or agricultural lands. Although the precise rates of tropical deforestation are a matter of debate, in Amazonia, Central America, and Indonesia, road construction projects, mineral development, and infrastructure projects since 1970 have led to initial human settlement and rates of deforestation of 0.25-1.0% per year (Curran et al., 1999; Turner et al., 2001; Moran et al., 2002). This tropical deforestation is particularly significant for biodiversity because these forests contain from one half to two-thirds of the world’s plant and animal species
(Wilson 1988; Reaka-Kudla 1997). In sub-Saharan Africa (Serneels and Lambin, 2001) and other parts of the developing world, cultures practicing low-intensity agriculture or nomadic pastoralism are being replaced by cultures using increasingly permanent and intense agriculture. And in the developed world, urban societies are growing and sprawling from cities to rural landscapes in pursuit of natural amenities (Hansen and Brown, in review). Exurban development (low-density homes, ca. 6-25/km$^2$) was the fastest growing land use type in the US since 1950 and now covers 25% of the area of the contiguous US (Brown et al. in review). Fueled by increasing human populations, technology, and wealth, these rapid changes over the last 30-50 years are heavily affecting the shrinking remaining natural habitats and reorganizing plant and animal communities in managed landscapes.

Figure 1. Typical trajectory of land use change. A given location may remain in wildland or proceed through one or more of the transitions. A regional landscape is typically a mosaic of all of these stages. From Hansen et al., in review.

The current wave of land use change is having strong effects on biodiversity for several reasons. First, new land cover conversion and intensified use of already converted lands are occurring globally, with especially fast rates in the tropics (Houghton 1994). Second, intact habitats were already reduced substantially in area and the current habitat conversion is pushing many ecosystems past thresholds of minimum size (Brooks et al., 1999a). Thirdly, while the global network of nature reserves has been expanded in recent decades (IUCN 1997), we are learning that land use change around even our largest reserves can negatively impact biodiversity and ecosystem processes within these reserves (Hansen and Rotella, 2002).

Our understanding of the locations, rates, and consequences of land use change has been greatly improved by application of satellite-based remote sensing and related global data sets. Accurate maps of the spatial distribution of land use were not widely available prior to the advent of earth-observing satellites in the 1970s. A variety of sensors are now in space collecting data about the earth at different temporal frequencies and spatial scales (Turner et al., 2003). Some of these sensors have now been in place long enough to allow quantification of change in land use based on comparisons of imagery recorded at two different time periods (Skole and Tucker, 1993; Steininger et al., 2001). These satellite data are increasingly used in conservation planning in conjunction with data on species distributions to prioritize locations in greatest need of conservation and to monitor important habitats (Steininger et al., 2001). When coupled with simulation models, satellite data can form the basis of projections of regional landscapes under various future management scenarios in order to assess the implications for biodiversity. The NASA Land Cover Land Use Change Program (LCLUC) (http://lcluc.gsfc.nasa.gov/) has been a major vehicle stimulating the use of satellite data for better understanding the rates and consequences of land use change.
The goal of this chapter is to synthesize current knowledge on the influences of recent land use change on biodiversity. We first review the ways by which land use change affects biodiversity. We then summarize the use of remote sensing for studying land use change and its influence on biodiversity. Then, drawing heavily on NASA LCLUC case studies, we examine the consequences of land use change for biodiversity. Finally, we highlight promising conservation and management efforts aimed at better sustaining biodiversity and human communities undergoing land use change.

2 Relevance of Land Cover and Use to Biodiversity

Of the multiple drivers of changes in biodiversity, land use change will likely have the largest effect on terrestrial ecosystems in the coming century (Sala et al., 2000). Changes in land cover and land use influence biodiversity by altering habitat, ecological processes, biotic interactions and human disturbance (Marzluff 2000; Hansen et al., in review) (Figure 2). These mechanisms act on the population dynamics of individual species via changes in rates of birth, death, and movements. The aggregate responses of individual species define patterns of community diversity.

Figure 2. The pathways by which land cover and land use influence biodiversity including the dynamics of species and the structure of communities. Data from remote sensing and other data (e.g., climate, human socioeconomics) provide a means of quantifying land cover and land use and understanding its impacts on biodiversity.

Perhaps the most obvious repercussions of land use change are loss, fragmentation, and degradation of habitat. Conversion of natural habitats to agriculture or other intensive human land uses causes these areas to become inhospitable for many native species. This conversion also reduces the area of natural habitats. Established theories in island biogeography (Rosenzweig 1995) and empirical evidence (Pimm and Askins, 1995; Brooks et al., 1999b) indicate that community diversity declines as habitat area is reduced as a function of the well-known species area relationship.
Smaller habitats can support fewer individuals within a population, hence rates of extinction increase with habitat loss (Pimm et al., 1988). The spatial pattern of habitat also influences biodiversity potential. Habitats with small patch sizes, increased edge to area ratios, and increased distance among patches fail to support habitat interior specialists and species with poor dispersal abilities (Laurance et al., 2002). Within forest stands, simplification of the number of canopy layers and other measures of forest structure reduces the microhabitats available to organisms and again, reduces biodiversity (Hunter 1999).

Human land use and habitat conversion together can have repercussions on other aspects of ecosystem dynamics including ecological processes. For example, the effects of land cover change on climate (Bonan et al., this volume) alter biodiversity depending on the sensitivity of species to climate change. Rainfall reductions in Costa Rican Monte Verde cloud forests resulting from land cover conversion in the valleys below have clear repercussions for the species relying on that habitat (Lawton et al., 2001). Natural disturbances such as wildfire are also altered by human land use. Consequently, species dependent upon early seral habitats may be lost if disturbance is suppressed or species specializing on late successional habitats may be lost if disturbance frequency and severity are increased (Huston 1994). Globally, land use is resulting in widespread changes in fire and flooding regimes with strong consequences for biodiversity. Although outside the focus of this chapter, the impacts on ecological processes due to land use are not only terrestrial. Freshwater systems are perhaps the most imperiled in the US and globally (Saunders et al., 2002). Land-use changes resulting in sediment loading and nitrogen fertilization constitute one of the chief threats to the health of these ecosystems. Watersheds transmit these detrimental effects further downstream into the coastal zone where enhanced fertilization may result in harmful algal blooms and fouling of coral reefs by sedimentation.

Some of the consequences of land use change are much less visible, because they involve not habitats but the organisms within habitats. Human activities often result in changed numbers and distributions of native species, as well as the introduction of alien species and pathogens. As a result, biotic interactions among species are changed, and ecosystem traits are altered. Among the biotic interactions affected are competition, predation, and disease and parasitism (Marzluff 2001). For example, the abundance of large carnivores tends to decrease near human development, resulting in mesopredator release in which smaller, edge-adapted predators increase in abundance (Soule et al., 1988). Herbivores are also released by the elimination of large predators in developed areas, resulting in increased herbivory and decreased plant diversity. Diseases may be spread among humans, domestic animals, and wildlife. Canine distemper virus, canine parvovirus and sarcoptic mange (Sarcoptes scabiei) are three pathogens known to have spread due to domestic dog-wildlife interactions, and are suspected to have caused population declines in the endangered gray wolf (Canis lupus) and black-footed ferret (Mustela nigripes) (Daszak et al., 2000). The introduction of exotic species is perhaps the most widespread type of biotic change resulting from land use. Human activities have allowed many species to overcome geographic boundaries and spread to new locations, sometimes with devastating effects on local species and ecological processes.

Humans also interact directly with native species through exploitation and inadvertent disturbance. In many parts of the world legal hunting or illegal poaching greatly reduce particular species of wildlife (Campbell and Hofer 1995; Escamilla et al.,
These activities, which can lead to “empty forests” and cascading effects on trophic structure, depend on human population densities as well as cultural characteristics, food preferences, and economic opportunities for the local human populations (Brashares et al., 2001). Outdoor recreation such as hiking and off-road vehicle use is increasingly popular in and around natural habitats and can act to displace wildlife, influencing reproduction, survival, and population dynamics (Miller et al., 2001; Fairbanks and Tullous, 2002). In many countries, there has been an increase in vehicle miles traveled per person and per household, escalating the potential for roadkill. Between 1980 and 2000, overall per capita vehicular travel in the United States increased by 48.7% (Charlier 2002). Roadkill has affected the demographics and migrations of birds, snakes, invertebrates, and amphibians, and is a major cause of mortality for moose, lynx, wolves, and American crocodile in various regions of the United States (Trombulak and Frissell, 2000). Finally, domestic pets displace and kill wildlife. For example, in Florida, pet dogs have affected the distribution of the endangered key deer, and are suspected to have eliminated them from several islands in the Florida Keys.

3 Use of Remote Sensing for Understanding Land Use and Biodiversity

Almost a century of development in biogeographic theory has promoted the tight connection between biodiversity and the condition of the surrounding habitat. The Arrhenius equation or power model of the 1920s made the basic mathematical linkage between the number of species and area (S=cA^z) (Meffe and Carroll, 1997). Given our increasing appreciation of a fundamental relationship between biodiversity and habitat, the ability of various remote sensing tools to detect habitat condition makes remote sensing an obvious choice for measuring and monitoring aspects of biodiversity (Figure 2). To a first approximation, remote sensing can tell us whether or not a particular habitat continues to exist. Thus, it can provide us with the area term in the Arrhenius equation. Remote sensing can also be used to quantify the degree of fragmentation of the habitat. With this information, one can estimate the number of species remaining within a habitat in question or, more accurately, likely to remain in that habitat after the effects of the habitat loss have worked their way through the ecosystem. Remote sensing also provides important information on the spatial and temporal distributions of other biophysical predictors of species distributions, including primary productivity, climate, and disturbances. Knowledge of land cover and other biophysical parameters serve as a proxy as indirect measures from which species distributions can be inferred from knowledge of habitat preferences (Turner et al., 2003). Here we describe land cover and other biophysical data from remote sensing, followed by emerging possibilities to obtain more direct measures of biodiversity.

3.1 LAND COVER FROM REMOTE SENSING

The last few decades have witnessed advances in the use of remote sensing to detect distributions of land cover at local, regional, and global scales. The impetus behind this research was primarily to improve land cover depictions in biosphere-atmosphere models simulating exchanges of energy, water, and momentum between the land surface and the atmosphere and in terrestrial ecosystem models simulating carbon
dynamics, rather than for biodiversity studies (Townshend et al., 1994). An important exception was the United States Geological Survey’s Gap Analysis Program, which uses Landsat data to map land cover throughout the United States for regional conservation assessments of native vertebrate species (Scott and Jennings, 1998). Here we describe these land cover products and their potential utility for biodiversity studies.

3.1.1 Global land cover products

Global-scale products generally describe the distributions of broadly-defined biomes, applicable for understanding biodiversity over the broad range of landscapes. The products derived in the 1980s and 90s used input data from the Advanced Very High Resolution Radiometer (AVHRR) on board the US National Oceanic and Atmospheric Administration (NOAA) series of meteorological satellites, at a spatial resolution of 8 km for Global Area Coverage (GAC) and 1 km for Local Area Coverage (LAC). GAC data is available for a time series beginning in 1982 (Agbu and James, 1994), while LAC data is only available globally for a few years in the 1990’s. The advantage of the coarse resolution data for land cover mapping lies in its global coverage and frequent daily acquisition, so that it is possible to derive composited products that preferentially select pixels with reduced cloud contamination. The availability of an annual time series of monthly-composited NDVI values made it possible to derive a number of global products including a coarse one-by-one degree resolution land cover map for application in biosphere-atmosphere models (DeFries and Townshend, 1994) and classifications at 8-km (DeFries et al., 1998) and 1-km (Hansen et al., 2000; Loveland et al., 2000) spatial resolutions. Recently, land cover products have been derived from the Moderate Resolution Imaging Spectroradiometer (MODIS) at 1-km spatial resolution (Friedl et al., 2002) and from SPOT Vegetation data (GLC 2000). These products indicate the broad patterns of biodiversity determined by the distributions of major biomes. With the exception of the GAC data analyzed in conjunction with higher resolution Landsat data (DeFries et al., 2002), the length of the time series has been too short and the spatial resolution too coarse to identify locations with threats to biodiversity from land cover change on a global scale.

3.1.2 Regional to landscape level land cover

The regional pattern of land cover is one of the determinants of species diversity across a landscape comprising more than one kind of natural community. Several efforts have applied successive time series of Landsat data to depict baseline land cover and changes in land cover from the 1970’s to present at a regional scale, mostly in tropical forest regions. Steininger et al. (2001) identified rates of change in Bolivia, Sader et al. (2001; 1994) in Central America, and the Landsat Pathfinder Project on Deforestation in the Humid Tropics (Skole and Tucker, 1993; Townshend et al., 1995; Kalluri et al., 2001) throughout the tropical forest belt. These analyses have led to identification of “hotspots” where deforestation is occurring. Limited acquisitions from the Landsat sensor, since improved with the launch of the Landsat Enhanced Thematic Mapper in 1999 (Goward and Williams, 1997), poses challenges to such historical analyses, as do the limitations imposed by the need for visual interpretation rather than automated analysis in hazy and poorly-calibrated scenes (Townshend et al., 1997). The Brazilian Space Agency (INPE) also uses Landsat data to annually map and report deforestation throughout the Brazilian Amazon (Houghton et al., 2000; INPE 2000).
New Landsat and SPOT imagery, along with imagery from Japan’s Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) instrument provide visible and near-infrared imagery at spatial resolutions of between 10 m and 30 m. Such imagery is useful for studies at regional scales, which typically require working with anywhere from 10 to 200 scenes. However, it is also applicable to work at landscape scales, which generally require on the order of one scene or a subset of a scene. The 30-year record of Landsat data (80 m resolution prior to 1984) and its utility at both regional and landscape scales will likely ensure the long-term continuation of 10 m to 30 m resolution satellite remote sensing.

3.1.3 Local-scale remote sensing
At a local-scale, vegetation structure and fine patterns of habitat distributions are important. Active remote sensing technologies (radars and lidars), which emit pulses of electromagnetic radiation and measure the return signal, have shown promise detecting vegetation structure at these finer scales (Lefsky et al., 2002). The Laser Vegetation Imaging Sensor (LVIS), an airborne waveform lidar, mapped canopy heights and topography to within 1 m and enabled estimations of vegetation density in the closed canopy tropical forest (Drake et al., 2002).

Local scale studies typically require remote sensing imagery at very high spatial resolutions. Traditionally, this has been the domain of aerial photography. However, two commercial satellites are now providing imagery at resolutions comparable to those of airborne systems. Space Imaging’s IKONOS satellite provides multispectral visible and near-infrared imagery at 4-m spatial resolution and panchromatic imagery at 1-m resolution. Digital Globe’s QuickBird sensor captures imagery at slightly higher spatial resolutions of 2.4 m multispectral and 0.6 m panchromatic. Satellite imagery at these scales allows the direct detection of individual trees and the detection of certain aspects of vegetation structure, which are relevant to biodiversity.

3.2 OTHER BIOPHYSICAL PREDICTORS OF BIODIVERSITY

In addition to land cover as an important determinant for biodiversity, several other related biophysical characteristics also influence species distributions, population sizes, and ranges.

3.2.1 Temporal vegetation dynamics
The seasonal distribution of primary production is an important determinant of population size and community richness, particularly in more arid settings where forage is a limiting factor (Huston 1994). The temporal patterns of the Normalized Difference Vegetation Index (NDVI) as observed with remote sensing sensors clearly reflect these patterns. Application of the monthly NDVI time series in terrestrial carbon models, along with other input data sets related to climate, soils, and land cover, provide information on the large-scale spatial and temporal patterns of primary production (Field et al., 1995). These distributions allow analyses of the movements, ranges, and population sizes of migratory species and how they are affected by spatial and temporal distributions of food resources.
3.2.2 Disturbance events
Remote sensing to identify disturbance events and recovery from disturbance, such as the fire product (Justice et al., 2002) and burn scars (Roy et al., 2002) from MODIS data, potentially provides insight into the repercussions for populations and communities. Habitat fragmentation exposes increased area to disturbance. In the Biological Dynamics of Forest Fragments Project, the largest experimental study of habitat fragmentation, Laurance et al. (2002) found that edge effects, such as susceptibility to fire, have substantial influences on habitat. These edge effects have multiple repercussions for biodiversity through altered species richness and abundances, species invasions, forest dynamics, trophic structure of communities, and a variety of other ecological and ecosystem processes.

3.2.3 Climate
Climate influences organisms directly and indirectly by its effects on primary productivity. While climate can be interpolated across landscapes where ground-based meteorological stations exist (Thornton et al., 1997), satellites offer hope of recording climate in more remote locations. The TRMM sensor acquires monthly mean hourly rainfall at a 1 degree x 1 degree spatial resolution. The new Advanced Microwave Scanning Radiometer (AMSR-E) sensor on NASA’s Aqua platform is designed to monitor soil moisture at a 1-km resolution.

3.3 OTHER HUMAN FACTORS
Habitat loss and fragmentation through land cover conversion, and to a lesser extent habitat degradation from land use change, are possible to examine from ground observations or remote sensing. Other types of land use, such as hunting and poaching, are more difficult to observe but nevertheless important for biodiversity (Campbell and Hofer, 1995; Escamilla et al., 2000; Revilla et al., 2001). These land uses, which can lead to “empty forests” and cascading effects on trophic structure, depend on human population densities as well as cultural characteristics, food preferences, and economic opportunities for the local human populations (Woodroffe and Ginsberg, 1998; Brashares et al., 2001). Although these factors are more difficult to observe and quantify, they are nevertheless essential to include in analyses about relationships between land use change and biodiversity.

4 Synthesis of Land Cover and Land Use Effects on Biodiversity
The availability of remotely sensed and other geographic data has greatly aided our understanding of the effects of land use change on organisms and communities. Maps depicting rates of change in land cover and use have stimulated several studies on the effects of both intensification in human dominated landscapes and of the consequences of this land use for the remaining natural habitats. These studies collectively reveal dramatic population expansions for human-adapted species and losses of species intolerant to human activities.
4.1 LAND USE INTENSIFICATION

As a landscape undergoes the trajectory of land use intensification outlined in Figure 1 the diversity of ecological communities typically drops. A general model of the spatial analog to this temporal trajectory was put forth by McKinney (2002). Along a spatial gradient from rural lands to urban cores, overall species richness decreases. The life histories and tolerances of species in the community underlie this relationship. While human-adapted species such as rock dove, and English sparrow, become more abundant with increasing land use intensity, many species decline due to loss of habitat, human disturbance, and biotic interactions with human-adapted species.

![Image of species richness distribution across land use gradient]

Figure 3. Distribution of species richness across a gradient in land use for studies of various organisms. Dashed lines represent unsampled portions of the gradient. Normalized species richness is calculated as a function of maximum number of recorded species at a point on the development gradient. From Hansen et al. in review. Studies are: insects – Denys and Schmidt (1998); bees – McIntyre and Hostetler (2001); birds – Blair (1996); lizards – Germaine et al. (1998); butterflies – Blair (1999); plants - Denys and Schmidt (1998).

Detailed studies along the land use gradient reveal that diversity sometimes peaks at intermediate land use intensities (Figure 3). This follows from the interplay of native and exotic species. While native species often decrease in diversity and abundance along the rural-urban gradient, the opposite is often true for non-native guilds. In a study around Tucson, AZ, for example, housing density best explained the increase in species richness for non-native birds (Germaine 1998). Similar patterns were documented for plant communities in Ohio as the percentage of non-native species increased along the rural-urban gradient (Whitney 1985). A synthesis of avian...
research found that over 90% of the surveyed studies documented an increase in exotic species with increasing settlement, while over 90% of the studies documented a decrease in interior nesters with increasing settlement (Marzluff 2001). The result is a non-linear response in which community richness and diversity peak at intermediate levels of development.

Research to date suggests that the types of species sensitive to land use change follows predictable patterns. As wildlands are converted to human land cover types, top-level predators are often the first species to decline. These species have large home ranges, low net reproductive rates, and are often dangerous to humans and livestock. Hence, these species are persecuted by initial settlers within natural habitats and slow to recover. In North America, the widely distributed grizzly bear and the wolf have been extirpated in all but the few remaining wilderness areas. Lions and other top predators in sub-Saharan Africa are also negatively correlated with human density (Woodroffe and Ginsberg, 1998).

As natural habitats are converted to agricultural and exurban uses, native species that are sensitive to predation often become reduced. This results because native meso-carnivores are released by the loss of top carnivores (Soule et al., 1988) and because nonnative predators may expand. In North America, neotropical migrant songbirds that use open cup nests are undergoing substantial declines, in part due to increased predation and nest parasitism (Terborgh 1988). On many Polynesian islands, native birds and mammals have been driven to extinction by exotic predators such as the brown tree snake and feral hogs (Diamond 2000).

Like mesopredators, some ungulate species thrive around intermediate levels of human land use. Several species of ungulates are positively associated with the settlements of traditional Masaii pastoralists lifestyle in the Massai Mara region of Kenya (Reid personal communication). In North America, white tailed deer, elk, and moose have expanded dramatically in recent decades in association with exurban and suburban development (e.g., Schneider and Wasel, 2000). This results both from reduced predation near human settlements and due to the increased primary productivity and forage production induced by domestic livestock and by lawns.

Species that are sensitive to the area of natural habitats may remain abundant in an increasingly human dominated landscape until native habitats are reduced below minimum size thresholds. Marzluff and Donnelly (in press) found in the forests around Seattle, WA, that native bird species richness remained high as forest area was reduced by development and dropped sharply where remaining forest was below about 15% of the landscape. In the transition from exurban and suburban to urban, many human adapted species drop out and overall species richness declines down to a relatively few urban exploiting species.

The trajectory toward more intense land use is not universal. In many developed countries, agricultural lands have been abandoned as industry has replaced agriculture as a primary economic driver (Huston in review). Consequently, forests and other natural cover types have expanded in places like the upper mid west and New England in the U.S (Mustard et al., this volume). Many native species are recovering in these landscapes, including some of the large predators that were extirpated early in initial EuroAmerican settlement.

The recovery of biodiversity in some human landscapes is instructive on the opportunity to achieve more sustainable land use. Within each of the land use types depicted in Figure 1, opportunities exist to better support native species while still
meeting human needs (McKinney 2002). Research and adaptive management to achieve ecological forestry, sustainable agriculture, and more nature-friendly cities are well underway.

4.2 REMAINING HABITATS

In human-dominated landscapes areas of natural habitat often remain. These may be designated nature reserves such as national parks or lands that have not yet been developed. These remaining natural habitats are extremely important for the conservation of biodiversity. We are increasingly learning, however, that these natural areas may be altered by human land uses in the surrounding lands (Figure 4). Land use in surrounding areas may affect ecosystem function and biodiversity within the natural habitats to greater or lesser extents depending on the size of the habitat, scaling of the surrounding ecosystem, and nature of human activities. The general mechanisms by which land use in the matrix may affect biodiversity within nature reserves were elucidated by Hansen and Rotella (2001) and Hansen and DeFries (in prep) (Table 1). These mechanisms also apply to natural habitats that are not formally protected as nature reserves.

Figure 4. Nature reserves and other natural habitats are often parts of larger ecosystems. Intense land use in the matrix surrounding the reserve may alter the properties of the surrounding ecosystem and influence the nature reserve under the mechanisms outlined in Table 1.

4.2.1 Habitat area

First, land use intensification reduces the functional size of natural habitats including the reserve itself and intact habitat in the surroundings. Reduction in functional size can simplify trophic structure, degrade the reserve’s ability to recover from natural disturbances, and increase species extinction rates. Trophic structure is altered as the area of natural habitats falls below the minimum needed for predators with large home ranges (Terborgh et al., 2001). Also, as natural habitat area decreases, the reserves may become too small for disturbance to maintain a dynamic steady state mosaic of seral stages (Pickett and Thompson, 1978).

As noted above, species extinctions occur as the area of natural habitat is reduced. The species area relationship has been used to predict the consequences of reducing the size of a habitat through conversion to intensive land uses (see Cowlishaw 1999 for a review). The species area relationship may be modified by several site-specific factors such as nonrandom associations between habitat destruction and spatial patterns of species richness, spatial aggregation of habitat fragmentation, and the home range requirements of organisms (Ney-Nifle and Mangel, 2000; Seabloom et al., 2002). DeFries et al. (in review) used this method to examine the potential loss of species from 198 protected areas in the world’s tropical forests. Over the last 20 years, 69% of
reserves in moist tropical forests experienced a decline in forest habitat within 50 km of their periphery, of which 37% respectively declined more than 5% (DeFries et al., in review). For those protected areas with changes in their effective areas (70%), capacity to conserve species richness averaged 93% relative to fully intact habitat in 1982, declining to 90% in 2001 with further decline to 51% in the hypothetical case of complete isolation.

Table 1. General mechanisms by which land use surrounding nature reserves may alter ecological processes and biodiversity within reserves. From Hansen and DeFries, in prep.

<table>
<thead>
<tr>
<th>Mechanism</th>
<th>Type</th>
<th>Description</th>
<th>Examples</th>
</tr>
</thead>
<tbody>
<tr>
<td>Change in effective size of reserve</td>
<td>Minimum Dynamic Area</td>
<td>Temporal stability of seral stages is a function of the area of the reserve relative to the size of natural disturbance.</td>
<td>Hurricanes in Puerto Rico (Shugart 1984)</td>
</tr>
<tr>
<td>Species Area Effect</td>
<td></td>
<td>As natural habitats in surrounding lands are destroyed, the functional size of the reserve is decreased and risk of extinction in the reserve is increased.</td>
<td>Fragmented forests in Kenya (Brooks et al., 1999b)</td>
</tr>
<tr>
<td>Trophic Structure</td>
<td></td>
<td>Characteristic spatial scales of organisms differ with trophic level such that organisms in higher levels are lost as ecosystems shrink.</td>
<td>Loss of predators on Barro Coronado Island (Terborgh et al., 2001)</td>
</tr>
<tr>
<td>Changes in ecological flows into and out of reserve</td>
<td>Initiation and runout zones</td>
<td>Key ecological processes move across landscapes. &quot;Initiation&quot; and &quot;run-out&quot; zones for disturbance may lie outside reserves.</td>
<td>Fire in Yellowstone National Park (Hansen and Rotella, 2001)</td>
</tr>
<tr>
<td>Location in air- or watershed</td>
<td></td>
<td>Land use in upper watersheds or air-sheds may alter flows into reserves lower in the water- or air-shed.</td>
<td>Rainfall in Monte Verde Cloud Forest. (Lawton et al., 2001)</td>
</tr>
<tr>
<td>Loss of crucial habitat outside of reserve</td>
<td>Ephemeral habitats</td>
<td>Lands outside of reserves may contain unique habitats that are required by organisms within reserves</td>
<td>Wildebeest in Serengeti National Park. (Seneels and Lambin 2001)</td>
</tr>
<tr>
<td>Dispersal/ Migration habitats</td>
<td></td>
<td>Organisms require corridors to disperse among reserves or to migrate from reserves to ephemeral habitats.</td>
<td>Elephant in East Africa (Cougenour et al., 2000)</td>
</tr>
<tr>
<td>Population source sink habitats</td>
<td></td>
<td>Unique habitats outside of reserves are “population” source areas required to maintain “sink” populations in reserves.</td>
<td>Birds around Yellowstone National Park (Hansen and Rotella 2002)</td>
</tr>
<tr>
<td>Increased exposure to humans at park edge</td>
<td>Edge effects</td>
<td>Negative human influences from the reserve periphery extend some distance into nature reserves.</td>
<td>Eurasian badgers in Donana Park (Revilla et al., 2001.)</td>
</tr>
</tbody>
</table>

4.2.2 Ecological flows

Land use change outside reserves can also alter flows of water and nutrients, climate, and disturbance agents into reserves. For example, biological reserves throughout the world are threatened by cumulative alterations in hydrologic connectivity within the greater landscape (Pringle 2000). Humans are altering hydrologic flows directly by dams, water diversions, ground water extraction, irrigation (Mustard this volume) and
indirectly by altering land cover which may change rates of transpiration, runoff, and soil storage. These flows of water transport energy, nutrients, sediments, and organisms. The location of a given reserve within a watershed, relative to regional aquifers and wind and precipitation patterns, can play a key role in its response to human disturbance transmitted through the hydrologic cycle. Reserves located in middle and lower watersheds often experience altered flow regimes, and inputs of exotic organisms and pollution from upstream. In contrast, reserves in upper watersheds may have intact physical habitat and contain important source populations of some native biota, yet human activities in lower watersheds may cause extirpation of species migrating from reserves, vectors for exotic species and disease to penetrate the upper watershed, and cause genetic isolation populations isolated in headwaters. This discussion of watershed effects also applies to airsheds. Regional land use change may alter climate and nutrient deposition within nature reserves considerable distances away (Lawton et al., 2001).

4.2.3 Crucial habitats
Reserves often do not contain the full range of habitats and conditions required by organisms. In this case, organisms may move outside of the reserve boundaries seasonally or during parts of their life histories to get access to crucial resources. If these crucial habitats outside of reserves are subjected to intense land use, populations of organisms within reserves may be reduced.

Figure 5. The Masaii Region of Kenya and Tanzania. Organisms such as wildebeest and elephant migrate outside of nature reserves such as Serengeti National Park seasonally. Replacement of nomadic pastoralism by crop agriculture and expansion of settlements has altered habitats outside of the nature reserves and resulted in substantial population declines of some migratory mammal species.

Ecosystems with high heterogeneity in climate and food resources are especially likely to have organisms that move long distances over the landscape to
acquire suitable resources. Populations of wildebeest in the Masai Mara portion of the Greater Serengeti Ecosystem of East Africa, for example, have declined dramatically, possibly due to the conversion of key seasonal habitat outside the reserve to commercial wheat farming (Serneels and Lambin, 2001) (Figure 5). Elephant, zebra, and other large mammals have also decreased as human settlements and croplands have expanded in this region (Coughenour et al., 2000). Similarly, in tropical forests of Borneo, Indonesia, long-distance migrations of bearded pigs have been disrupted by the logging of the diptocarp trees whose fruits are prime food sources for the pigs (Curran personal communication).

The crucial habitats outside of reserves may be especially rich in resources and act as population “source” areas. These habitats may allow subpopulations to produce surplus offspring that disperse to less-rich habitats in nature reserves and allow persistence of the subpopulations in the reserves. For example, in the Greater Yellowstone Ecosystem, Hansen and Rotella (2002) found that bird populations were concentrated in small "hot spots" in productive, lowland settings outside protected areas (Figure 6). Intense land use (exurban development) has converted these low elevation population source areas to sink areas and reduced the viability of subpopulations in the more marginal habitats in protected areas.

4.2.4 Edge effects
Human presence on the periphery of reserves may cause changes in ecosystem processes and biodiversity that extend varying distances into the reserve. Examples include hunting and poaching effects within reserves (Campbell and Hofer, 1995; Escamilla et al., 2000; Revilla et al., 2001). Other types of edge effects involve...
ecological processes such as disturbance rates and microclimate changes, human settlement and recreation, and introduction of exotic organisms and diseases. Many of these edge effects are proportional to the density of the adjacent human population (Woodroffe and Ginsberg, 1998; Brashares et al., 2001). Hence, these effects may be increased under human population growth around reserves.

In sum, myriad studies indicate that land use change is the primary driver of change in biodiversity across many portions of the globe. Natural habitats have been converted to more intense human land uses, with dramatic effects on native species and communities. Even the remaining natural habitats are not immune from the effects of land use change. Human activities in the matrix around natural habitats can alter ecological processes and organisms within the reserves. These findings suggest that the future ability of protected areas to maintain current species richness depends on integrating reserve management with regional land-use activities.

4.3 OVERLAP OF LAND USE AND BIODIVERSITY

The locations of intense human land use and areas rich in biodiversity are not random relative to one another. Rather, land use and hot spots for biodiversity often overlap. This is because both are influenced by biophysical factors such as climate and ecosystem productivity. A consequence of this overlap is that land use has a particularly strong influence on biodiversity.

The spatial distribution of biodiversity is influenced by many factors (Huston 1994). However, a dominant factor is available energy, represented both climate and as plant productivity (Huston 1994; Rosenzwieg 1994). Across North America, potential evapotranspiration explained more than 90% of the variation in vertebrate species richness (Currie 1991). And in sub-Saharan Africa, net primary productivity (NPP) accounted for 73% of the variation in vertebrate species richness (Balmford et al., 2001). In relatively unproductive ecosystems, species richness is generally positively related to ecosystem energy because larger population sizes can be supported, reducing the risk of extinction due to small population sizes (Pimm et al., 1988). In highly productive ecosystems, a few species may dominate the available resources and out compete other species, causing species richness to change negatively related to energy level. Hence, globally, species richness often peaks in locations that are intermediate in energy levels (Waide 1999).

The biophysical factors that influence ecosystem energy also influence human land use. Globally, human population density and intense human land use are greatest in locations that are either intermediate (Balmford et al., 2001) or high (Huston 1993) in ecosystem productivity. One reason for this pattern is that human populations are most dense on lands suitable for agriculture or in the low elevation and coastal areas that facilitate transportation (Huston, in review). Importantly, nature reserves, areas designated primarily to protect nature are biased in location to the less productive landscape settings that are lower in biodiversity (Scott et al., 2001).

The overlap between intense human land use and biodiversity is documented at a hierarchy of spatial scales. Globally, human density and population growth rate are disproportionately high near the world’s 25 richest biodiversity hot spots (Cincotta et al., 2000). Across the African continent, human density and vertebrate richness are significantly correlated (Balmford et al., 2001). Within the state of California, native plants and human density are both disproportionately high in coastal areas. Within the
Greater Yellowstone Ecosystem, native birds and rural home development are concentrated in the same small percentage of the landscape with mesic climate, good soils, and high NPP (Figure 6) (Hansen et al., 2002). The important conclusion is that at many spatial scales, humans and biodiversity tend to be concentrated in the same locations. Consequently, the human conversion of natural habitats and the effects on native species described are occurring most rapidly in the locations most important to biodiversity. This is resulting in elevated levels of species extinction in these locations (Kerr and Currie, 1995; Rivard et al., 2000; Brashares et al., 2001; Parks and Harcourt, 2002).

5 Conservation and Management

Effective management for biodiversity in the face of land use change is a tremendous challenge in that it requires expertise ranging across the physical sciences, life sciences, and social sciences. It also requires management across spatial scales from global to local. Managing biodiversity at the global scale would appear to be well beyond our grasp due to fundamental gaps in our knowledge, such as the number of species on the planet, and a lack of the necessary technical tools and political/administrative frameworks in which managers could hope to accomplish such a lofty goal. And yet, there are a surprising number of groups engaged in assembling the mix of knowledge, tools, and political structures necessary to achieve this goal.

First, managers have identified the major threats to biodiversity operating across spatial scales. As noted already, there is widespread consensus that habitat degradation and loss, resulting from changes in land use, constitute a primary cause – if not the primary cause – of biodiversity loss. Second, managers have developed a body of theory connecting island biogeography with landscape ecology’s emphases on the importance of habitat connectivity, edge effects, and the quality of the landscape matrix surrounding a crucial habitat; and the resulting approach is showing some predictive power. We have learned of the basic mechanisms by which increasingly intense land use alters ecological communities and processes. We have also learned how these effects can reach across the landscape and alter biodiversity even in protected areas.

Using data on the populations of endangered and threatened species from World Conservation Union (IUCN) Red Lists and historical sources in conjunction with information on the extent of habitat loss from remote sensing satellites and in situ measures, the theory of island biogeography has given managers the ability to predict likely species losses resulting from a given amount of habitat loss at regional and global scales (Wilcox 1980; Newmark 1995; Brooks et al., 1997; Brooks et al. 1999a; Brooks et al., 1999b; Brooks et al., 2002). Improved understanding of habitat connectivity, edge effects, matrix quality, and flows of energy and nutrients should enable us to refine these admittedly rough regional and global estimates to produce better projections for species losses at global, landscape and local scales.

Third, managers know, at a regional level, where to focus their efforts to conserve and then manage biodiversity. Not all areas are of equal importance. We have learned that some locations represent crucial habitat, migration corridors, and population source areas for populations occupying the broader landscape. We increasingly appreciate that ecological processes operate in space such that there are key initiation zones and run-out zones. We have found that species are not evenly
distributed across landscapes and regions, but are often concentrated in hot spots. Moreover, these hotspots for biodiversity also may overlap with locations of intense human land use. Driven by the need to find the best use for limited conservation resources, biogeographers and field ecologists have expended much effort over the past 5 to 10 years developing approaches to prioritizing the regions of the globe most important for biodiversity based on estimates of species richness, rarity and, in some cases, levels of endemism and/or threat. Examples include biodiversity hotspots, the global 200 ecoregions, endemic bird areas, and frontier forests (Bryant et al., 1997; Olson and Dinerstein, 1998; Stattersfield et al., 1998; and Myers et al., 2000). Thanks to these efforts we now have regional scale knowledge of areas that are critically important for maintaining the future of biodiversity.

Fourth, managers are just beginning to get access to tools that should allow them to monitor and assess over time the condition of these important regions. At global and regional scales, satellite remote sensing is perhaps the most useful tool for this purpose. Global, routine monitoring via satellite of disturbance events driving land-use change is still in its infancy but progress is being made. The MODIS sensors on the TERRA and AQUA satellites are providing fire products at 1-km spatial resolution on time scales ranging from daily to monthly (Justice et al., 2002). Other MODIS data products still under development include a quarterly 1-km land-cover change product and a monthly 250-m vegetative cover conversion product (Zhan et al., 2000). These routine products will provide a coarse-resolution early warning system for changes in land cover, which can then be viewed with higher spatial resolution satellite imagery or in situ devices. Newly assembled global Landsat data sets from the mid-1970s, circa 1990, and 2000 time periods will provide baseline moderate to high spatial resolution imagery (80 m for the 1970s, 30 m for circa 1990, and 30 m multispectral/15 m panchromatic imagery for 2000) from three decades to look at landscape scale change over time.

Despite this focus on satellite remote sensing, in situ sampling strategies are critical. Without in situ data, there is no reliable means of validating the satellite imagery. More importantly though, ground surveys will always be necessary to measure and monitor many elements of biodiversity. Remote sensing and the ecological models capable of ingesting it are only tools that allow a synoptic view of certain environmental parameters relevant to biodiversity, e.g., land cover.

This combination of theory, knowledge of regions important to biodiversity, and tools to monitor these regions over time provides the basis for a global system to manage biodiversity. What is lacking are the political and administrative structures to address biodiversity conservation and land-use change at broad scales. Here too, recent developments are encouraging.

Island biogeography theory and landscape ecology have driven managers to think big. For example, protected area managers must now look beyond the boundaries of their park at the surrounding matrix if they are to maintain the viability of the protected area and its biodiversity over time. As a result, national governments are beginning to explore mechanisms to manage ecological regions across national boundaries. Examples of transboundary conservation initiatives at various levels of integration include the Mesoamerican Biological Corridor, the Greater Serengeti Ecosystem, and the proposed Yellowstone to Yukon Conservation Initiative. The Central American heads of state instituted the Mesoamerican Biological Corridor (MBC) in 1997 to serve as a basis for joint planning among the seven nations of Central
America and 5 southern Mexican states. It addresses a regional commitment to sustainable development. The MBC grew out of an earlier plan to link a series of protected areas spread throughout the isthmus in a corridor known as *Paseo Pantera* or “path of the panther” (Miller et al., 2001). A desire to provide suitable habitat to sustain viable populations of keystone species, such as large predators, or critical ecological processes, e.g., migration corridors for large ungulates, have often provided the initial incentive for transboundary initiatives.

The Greater Serengeti Ecosystem consists of a series of protected areas on both sides of the Kenya/Tanzania border that maintain one of the world’s greatest migrations of large mammals (Miller 1996). The Yellowstone to Yukon Conservation Initiative is still in the development phase. It would link protected areas and surrounding lands in the Northern Rocky Mountains to protect biological diversity and the wilderness character of the region. It also drew inspiration from a large predator, in this case a female gray wolf that was tracked moving across 840 km over 18 months (Chester 2003).

The growing number of conservation initiatives crossing political boundaries provides a variety of political and administrative frameworks for the implementation of regional land-use planning in support of biodiversity. In this case, theory, tools, and practice appear to be coming together at the right time for integrated management. Of course, most decisions regarding land use, and the accompanying impacts on biodiversity, will continue to be local in scale.

It is perhaps at the local level where people can most easily grasp the influence of land use on biodiversity and see the benefits of conservation. Sustainable forestry protocols are increasingly guiding management practices within forest stands. Encouraged by market forces, many forest products companies globally are developing and implementing practices to maintain biodiversity values within the stands and landscapes that are used for wood and fiber production. Private citizens and non-governmental organizations are increasingly purchasing development rights on crucial habitats that are not currently protected. Such conservation easements allow traditional livestock grazing or agriculture that are consistent with maintenance of biodiversity while prohibiting more intense land uses such as subdivision. Individual citizens are increasingly aware of the ecological consequences of their actions and are modifying their lifestyles to reduce their ecological impacts. In the American west, for example, rural homeowners are learning how to live more lightly on the land through managing hobby livestock, weeds, and reducing supplemental foods to native wildlife. Increasingly, local communities are integrating consideration of biodiversity into land use planning, allowing more integrated landscape management across private and public lands. These local initiatives are partially motivated by the growing realization that good conservation has strong economic and social benefits (Rasker and Hansen, 2000). Perhaps our increasing appreciation of the interdependence between healthy ecosystems and healthy human communities will lead toward more sustainable approaches for land use.

### 6 Conclusions

Remotely-sensed data derived from satellites is providing a basis for understanding rates of land use change and consequences for biodiversity. We are now coming to
understand the global extent of the conversion of natural habitats to human-dominated landscapes. Recent decades have seen dramatic decreases in the remaining natural habitats and increases in the intensification of human land use throughout the tropical and temperate zones of the world. Within our managed landscapes, native species are giving way to exotic species and ecological communities are becoming increasingly homogenized. These effects of intense land use reach even into nature reserves and other habitats that appear to be healthy at a distance, but are undergoing important ecological degradation. Globally, many native species have gone extinct and past land use change has produced an “extinction debt” that will continue to be felt in future decades and centuries. Increasingly, we are realizing the value of biodiversity to human societies and economic systems. Hence, a myriad of efforts are underway from local to global levels to manage land use to better achieve human and natural resource objectives. Our challenge is to integrate remote sensing, scientific research, landscape management, and personal lifestyles to better sustain ecological and human communities.

Acknowledgements. We thank John Mustard, Heather Rustigan, and Scott Powell for helpful comments on the manuscript. Heather Rustigan prepared figures 5 and 6. Support was provided by the NASA LCLUC Program.

7 References


CHAPTER 17

LAND USE AND CLIMATE

GORDON B. BONAN1, RUTH S. DEFRIES2, MICHAEL T. COE3, DENNIS S. OJIMA4

1National Center for Atmospheric Research, P.O. Box 3000, Boulder, CO 80307 USA
2Department of Geography, University of Maryland, College Park, MD 20742 USA
3Center for Sustainability and the Global Environment, Gaylord Nelson Institute for Environmental Studies, University of Wisconsin-Madison, 1710 University Ave, Madison, WI 53726 USA
4Natural Resource Ecology Laboratory, NESB, B229, Colorado State University, Fort Collins, CO 80523 USA

1 Introduction

Terrestrial ecosystems affect climate through exchanges of energy, water, momentum, mineral aerosols, CO2, and other atmospheric gases. Changes in community composition and ecosystem structure alter these exchanges and in doing so alter surface energy fluxes, the hydrologic cycle, and biogeochemical cycles. As a result, changes in land cover through natural vegetation dynamics or human uses of land can alter climate.

Much of our knowledge of the influence of vegetation on global and regional climate comes from climate models. In these models, the absorption of radiation at the surface, the exchanges of sensible and latent heat between land and atmosphere, storage of heat in soil, and the frictional drag of the surface on wind influence climate. Important surface properties that determine these exchanges include: albedo, which determines the absorption of solar radiation at the surface; surface roughness, which affects turbulence and the turbulent fluxes of sensible heat, latent heat, and momentum; soil water, which affects the partitioning of net radiation into sensible and latent heat; vegetation, which alters the hydrologic cycle and also affects albedo, surface roughness, canopy physiology, and the leaf area from which heat and moisture are exchanged with the atmosphere; and soil texture, which affects infiltration, runoff, and soil water. The first generation of land models coupled to atmospheric models parameterized these processes using simple aerodynamic bulk transfer equations and simple prescriptions of albedo, surface roughness, and soil water (Sellers et al., 1997). These models evolved into a second generation of models such as the Biosphere-Atmosphere Transfer Scheme (BATS, Dickinson et al., 1993) or Simple Biosphere Model (SiB, Sellers et al., 1986) that include the full hydrologic cycle and vegetation effects on energy and water fluxes.

Simulation of the surface energy balance is a fundamental component of land models (Figure 1). Solar radiation is absorbed, reflected, or transmitted by foliage and woody material in the plant canopy. Longwave radiation is gained from the atmosphere and also emitted by surface elements. The absorbed solar radiation and net longwave radiation comprise the net radiation available at the surface. Some of this energy is dissipated as sensible heat, which warms the overlying atmospheric column, or as latent heat, which cools and moistens the atmosphere. These fluxes are inversely proportional to an aerodynamic resistance that parameterizes turbulent processes and vegetation re-
sistances that regulate energy and water transfer between leaves and surrounding air. Energy not dissipated as sensible or latent heat is used to melt snow or stored in soil.

![Diagram of surface energy fluxes and hydrology in land models.](image)

Figure 1. Surface energy fluxes (left) and hydrology (right) as represented in land models.

Water stored in the canopy, on the ground as snow, or in soil influences surface energy fluxes, and therefore land models also simulate the hydrologic cycle (Figure 1). Precipitation over land is stored in soil, returned to the atmosphere during evapotranspiration, or runs off to the oceans. Over long times scales (e.g., annually), changes in water storage are small so that runoff is the difference between precipitation input and evapotranspiration loss. The water cycle involves much more detail than this simple water balance. Some precipitation is intercepted by leaves, twigs, and branches and readily evaporates. The water that is not intercepted falls to the ground. Some of the water reaching the ground infiltrates into the soil. The remainder is either stored in small depressions or runs off over the surface. Soil water is returned to the atmosphere through evaporation from soil and transpiration from plants.

Vegetation is an important determinant of energy fluxes and the hydrologic cycle. Land models typically represent vegetation structure in terms of height, leaf area index and its phenology, and root profile in soil. The physiology of vegetation is represented by vegetation types that vary in leaf optical properties and stomatal physiology. A third generation of models relates photosynthesis, transpiration, and stomatal conductance to provide a consistent parameterization of energy, water, and carbon exchanges (Sellers et al., 1997). A fourth generation of models currently under development allows vegetation to change as climate changes (Foley et al., 1998, 2000).

Land management practices such as grazing, forestry, and conversion to arable lands affect trace gas fluxes. Changes in land management have contributed to decreased CH$_4$ oxidation, increased CO$_2$ emission, and N$_2$O production from soils. CO$_2$ emissions resulting from land use change since the 1980’s are approximately 40% of fossil fuel contributions. Use of nitrogen fertilizers and increased atmospheric loading of nitrogen have contributed to increased CH$_4$ and N$_2$O in the atmosphere (Mosier et
In addition, increased use of nitrogen has altered community composition and ecosystem functions (Vitousek et al., 1997).

Presently, a large portion of the natural vegetation of the world has been cleared to grow crops. By changing net radiation at the surface, the partitioning of this energy into sensible and latent heat, and the partitioning of precipitation into runoff and evapotranspiration, changes in land cover can alter climate. The impact of past land cover change on climate is noticeable at the global scale (Hansen et al., 1998; Brovkin et al., 1999; Govindasamy et al. 2001; Zhao et al., 2001; Bounoua et al., 2002). Land cover changes over the next 50 years due to human land uses are also likely to alter climate, especially in the tropics, subtropics, and semiarid land (DeFries et al., 2002a). In the remainder of this chapter, we review the climatic impacts of land cover change in tropical forests, temperate forests and grasslands, boreal forests, and drylands.

2 Tropical Deforestation

Studies of the micrometeorology of forested and pasture sites in Amazonia show some of the climatic effects of tropical deforestation (Gash et al., 1996; Gash and Nobre, 1997). Pastures have a higher albedo than forests, are warmer, and emit more longwave radiation. Consequently, net radiation for pastures is less than for forests. Another important difference is that pastures have a lower aerodynamic roughness than forests. During the rainy season, evapotranspiration from pastures is typically less than from forests due to the reduced available energy and reduced roughness. During the dry season, shallow-rooted pastures tend to have low evapotranspiration as the surface soil water is depleted. In contrast, forests have no significant reduction in evapotranspiration as the deep-rooted trees extract water from deep soil. The deep roots of trees may be a key control of climate, creating a cooler, moister climate because of sustained transpiration during the dry season (Kleidon and Heimann, 2000). The net result of these differences is that pastures are warmer during the day than forests. The largest difference occurs during the dry season, when evapotranspiration is reduced in pastures.

Climate model studies have examined the impact of topical deforestation on climate, especially in Amazonia (Dickinson and Henderson-Sellers, 1988; Nobre et al., 1991; Henderson-Sellers et al., 1993; Zhang et al., 1996a,b; Costa and Foley, 2000; Bounoua et al., 2002). Most studies find that complete transformation of forest to pasture results in a warmer surface with decreased evapotranspiration and precipitation. These studies show that large-scale tropical deforestation over regions greater than 100,000 km² is associated with a simulated decrease in the total atmospheric column heating (through an increase in surface albedo and decrease in net radiation) and a subsequent decrease in the atmospheric vertical motion, cloudiness, and rainfall over the deforested region. Delire et al. (2001) showed that in Indonesia ocean dynamics produces further feedback. Windspeeds over the adjacent ocean are increased due to decreased surface roughness over the deforested land. This leads to an increase in the ocean upwelling, a decrease in the total atmospheric column temperature, and a further decrease in the regional atmospheric convection and precipitation.

Observational evidence for the large-scale feedback between land cover change and precipitation is limited. Pielke et al. (1998) analyzed GOES satellite data and found that with tropical deforestation at scales of thousands of kilometers or more alterations in the cloud patterns (and presumably precipitation) are observed. In mon-
tane environments, feedbacks between deforestation and cloud formation have been observed to occur at smaller spatial scales (Lawton et al., 2001).

In addition to affecting precipitation, deforestation alters runoff. Observational evidence suggests that direct impacts of land cover change on hydrology are important at spatial scales ranging from to 10 to 100,000 km$^2$. Studies of small watersheds in tropical, temperate, and boreal regions show that water yield and discharge generally increase with increasing deforestation and decrease with reforestation (Bruijnzeel, 1990; Sahin and Hall, 1996). Greatest water yield increase occurs after more than 20% of the basin is deforested. Confirmation of this process in basins larger than 100 km$^2$ is rarely possible because of the logistics of monitoring very large watersheds. However, Costa et al. (in review) used a 50 year record of river discharge and precipitation in conjunction with land use census data to assess the impact of rapid land cover changes on the discharge of the 176,000 km$^2$ Tocantins River basin of northern Brazil. Their analysis shows that a rapid increase in land used for agriculture (from 30% of the basin in 1960 to 50% in 1995) is coincident with a 25% increase in the discharge of the Tocantins River despite no significant change in precipitation. The land cover change appears to have reduced evapotranspiration and increased runoff and discharge.

Changes to the regional water budget due to feedbacks between land cover change and the atmosphere dominate only at large spatial scales (100,000 km$^2$). At small spatial scales changes in the surface energy, moisture, and momentum budgets are insufficient to cause noticeable feedbacks to regional circulation. Evapotranspiration decreases as does total atmospheric heating, but not enough to impact regional precipitation. However, with increasing scale of deforestation the impact of decreased evapotranspiration and increased surface albedo may eventually lead to a reversal of the direct impact and reduction in precipitation and runoff. For regions of large, fairly homogenous vegetation such as tropical rainforests, the spatial scale for feedbacks to occur is large (100,000 km$^2$). However, for particular settings such as mountainous islands or isolated mountains on land, deforestation on small scales (100 km$^2$) can lead to significant local reduction in precipitation and runoff (Lawton et al., 2001).

Land clearing in many of these tropical regions is associated with biomass burning. Biomass burning releases large amounts of CO$_2$, aerosols, and reactive nitrogen species to the atmosphere. These emissions contribute to the recent changes in greenhouse gas fluxes from agricultural lands.

3 Temperate Land Use

Much of the natural vegetation of Europe has been cleared for cropland. The Mediterranean region of southern Europe and northern Africa has received particular attention because of extensive deforestation during the past 2000 years. Climate model simulations suggest that this deforestation contributed to the dryness of the current climate (Reale and Shukla, 2000; Heck et al., 2001). For example, simulations by Reale and Shukla (2000) show that decreased surface albedo with natural vegetation increases precipitation by strengthening the land-sea temperature contrast and altering atmospheric circulation over northern Africa and the Mediterranean Sea. This results in a northward shift in the intertropical convergence zone and a local circulation between the Mediterranean Sea and northwestern Africa in the region of the Atlas Mountains.
Results are less pronounced and more variable in southern Europe. The climate of southern Europe is especially sensitive to changes in leaf area index and rooting depth (Heck et al., 2001). Replacement of forests by cropland and grassland has greatly reduced leaf area index and rooting depth compared to natural vegetation. The greater leaf area and deeper rooting depth of natural vegetation leads to a moister and cooler spring followed by a warmer and drier summer compared to present vegetation. Evapotranspiration increases from April until mid-July, cooling the surface, moistening the boundary layer, and enhancing precipitation. Thereafter, soil water limits evapotranspiration and the climate sensitivity to natural vegetation is reversed.

The forests of eastern U.S. and much of the grasslands in the Great Plains of central U.S. have been replaced with crops. Possible climate effects of this deforestation were a concern early in the settlement of North America (Bonan, 2002). Climate model simulations have begun to document the impact of land use on the climate of the U.S. (Bonan, 1997, 1999; Hansen et al., 1998; Bounoua et al., 2002). These studies demonstrate that clearing of forests in eastern U.S. for agricultural has cooled climate. The cooling largely arises from increased surface albedo that decreases the net radiation at the surface and increases in latent heat flux during the growing season. The cooling is larger for daily maximum temperature than for daily minimum temperature so that diurnal temperature range decreases.

Settlement of the U.S. Central Plains in the 1870s and 1880s shifted scientific and popular interest in the effects of land use on the climate from deforestation to cultivation and tree planting. In particular, there was a widely held belief that land clearing, cultivation, and tree planting would increase rainfall in the semi-arid climate of the Great Plains (Bonan, 2002). The climatic effect of converting natural grasslands in the Great Plains to croplands has been studied with regional climate models (Eastman et al., 2001). This study shows that introduction of croplands has warmed daily maximum temperature by a few degrees over the growing season throughout most of the Central Plains. Seasonally varying differences in surface albedo and sensible and latent heat between grasslands and croplands control this warming. Early in the growing season, the albedo of bare cropland soil is lower than that of grassland. Differences in albedo decrease as the crops grow, but increase again with harvesting. Differences in sensible and latent heat fluxes have an important role in the warming. The warming from croplands is weak early in the growing season because of high evaporation from bare fields saturated with soil water. Later in the growing season, the leaf area index of croplands is less than that of grasslands, leading to less transpiration, more sensible heat, and higher daytime temperatures.

4 Boreal Forest-tundra Ecotone

The boreal forest is the northernmost forest, lying just south of the treeless tundra. Because of differences between boreal forest and tundra ecosystems in albedo, roughness, and the partitioning of energy into latent and sensible heat, the geographic extent of these ecosystems is an important regulator of global climate. Numerous climate model studies have found that the presence of the boreal forest warms climate compared to tundra (Bonan et al., 1992; Thomas and Rowntree, 1992; Chalita and Le Treut, 1994; Foley et al., 1994; Douville and Royer, 1996). One important difference between forest and tundra is surface albedo when the ground is covered by snow. Trees protrude over
snow and mask its high albedo. As a result, treeless areas have a higher albedo when snow is on the ground than do forests (Baldocchi et al., 2000). Forest and tundra ecosystems also differ in how they partition net radiation into sensible and latent heat fluxes (Eugster et al., 2000). Throughout summer, tundra sites generally have higher latent heat flux and lower sensible heat flux compared to nearby needleleaf evergreen forests. The low evaporative fraction of needleleaf evergreen forests arises in part from low stomatal conductance due to low foliage nitrogen content and low photosynthetic capacity (Baldocchi et al., 2000; Eugster et al., 2000).

The forest-tundra ecotone may control the summer position of the Arctic front in North America due to the contrast in albedo, roughness, and energy exchange, which results in strong heating of the atmosphere over forest and weak heating over tundra (Pielke and Vidale, 1995; Lynch et al., 2002). This creates a warm, dry surface climate and a deep planetary boundary layer during the course of a typical summer day. The deep boundary layer may feed back to limit evapotranspiration by entraining a large amount of dry air, which decreases stomatal conductance by increasing the vapor pressure deficit and forcing stomata to close (Baldocchi et al., 2000; Eugster et al., 2000).

Paleoclimate studies suggest that the forest-tundra ecotone has a large role in regulating climate. Expansion of tundra at the expense of forest may have played a role in the onset of glaciation (Gallimore and Kutzbach, 1996; de Noblet et al., 1996). Climate model simulations also show that reduction in forest cover at middle to high latitudes at the last glacial maximum reinforced the cold climate (Levis et al., 1999). As climate warmed and the glaciers retreated northwards, the treeline migrated northwards. The decrease in surface albedo caused by the northward expansion of the boreal forest in response to the climate warming accentuated the warming (Foley et al., 1994).

5 Dry Lands

In arid and semi-arid climates, overgrazing of rangelands by livestock, expansion of agriculture, and increased fuelwood collection can alter the surface energy balance and hydrologic cycle and thereby change climate. Many climate model studies, beginning with the studies of Charney (1975) and Charney et al. (1977), show that conversion of natural vegetation to managed systems causes a feedback with the atmosphere that reduces precipitation. In addition, observational studies suggest that at relatively small scales the conversion from natural vegetation to a managed system with reduced rooting depth and leaf area index results in decreased evapotranspiration and greater water yield (Sahin and Hall, 1996).

The Sahel region of northern Africa (15˚W to 20˚E longitude, 13˚N to 20˚N latitude) has been a region of intense investigation of land surface-atmosphere feedbacks. Persistent drought has affected the Sahel since 1969. Almost every year since 1969 has been dryer than the average for the period 1931-1960 and the mean precipitation is about 25-40% less than the earlier period (Nicholson, 2000). Charney (1975) suggested that higher albedo from overgrazing was responsible, at least in part, for the persistent drought. Decreased net radiation at the surface cools the atmosphere, which promotes subsidence of air aloft. Subsidence decreases cloud formation and convection, leading to less rainfall. Subsequent studies have generally concluded that very large changes in land cover in the Sahel could cause a significant drought in the region (Xue
and Shukla, 1993; Xue, 1997; Clark et al., 2001). However, the land cover changes required to simulate a climate change as large as observed are much greater than those that are observed. Therefore, the cause of the climate shift to dry conditions in the Sahel is believed to be more complex (Foley et al., in press).

Wang and Eltahir (2000a,b) indicate that the Sahel may have two stable states: a wet phase and a dry phase. Strong feedbacks between vegetation, the surface energy and water balance, and the West African monsoon appear to be responsible for the existence of these alternative steady states. Slowly changing land cover or sea surface temperatures may act as the trigger for a rapid climate transition, resulting in a feedback to the climate. Once the climate shift has occurred, the feedbacks then maintain the system in that regime for decades until some other triggering event forces climate into the other regime (Foley et al., in press). In this theory, the drought of the last 30 years could have been triggered by an initial degradation of as little as 20% of the natural vegetation of the Sahel or by a temporary (2 year) warming of Atlantic sea surface temperatures.

Studies in the Sonoran and Negev deserts show the effects of overgrazing on local-to-regional climate. Overgrazing in the Sonoran desert in Mexico has reduced vegetation cover, exposed soil, and increased surface albedo. Although the higher albedo of the overgrazed land should cool the surface, lower vegetation cover reduces evapotranspiration and warms the surface (Bryant et al., 1990; Balling et al., 1998). In addition, sparse vegetation and compacted soils result in less infiltration, more runoff, and drier soils that further reduce evapotranspiration. Overgrazing and fuel wood collection have increased surface albedo in the Negev desert (Otterman, 1989). Because of higher albedo, overgrazed soils are cooler than vegetated areas. These studies show the climatic effect of overgrazing drylands depends on soil moisture (Bonan, 2002). In both regions, overgrazing increases surface albedo. In the dry soils of the Negev, albedo is more important than soil moisture and overgrazing leads to cooling. In less arid regions such as the Sonoran, soil moisture prevails over albedo. The lower evapotranspiration of overgrazed land warms the surface despite higher albedo.

Changes in land cover alter the hydrology of drylands. Tree clearing in the Murray-Darling basin of Australia is thought to have decreased evapotranspiration and raised the water table (Pierce et al., 1993). Lack of rainfall to grow crops is also a problem in drylands. Irrigation is needed to augment sparse rainfall. The large contrast in sensible and latent heat between irrigated croplands and surrounding dry vegetation can generate mesoscale circulations (Anthes, 1984; Avisser and Pielke, 1989; Chen and Avisser, 1994). The high evapotranspiration from irrigated soils moistens and cools the atmosphere near the surface, creating circulations akin to sea breezes between the cool, wet agricultural land and hot, dry native vegetation. Irrigation is likely creating a cooler, moister climate in northeastern Colorado (Stohlgren et al., 1998; Chase et al., 1999). In addition, irrigation has reduced the discharge of many rivers, drained many inland water bodies, depleted groundwater reservoirs, and salted soil with runoff.

Land use change in arid and semiarid regions alters the biogeochemical fluxes of soot, nitrogen, carbon compounds (e.g., CO, CO$_2$, volatile organic compounds), and aerosols. Land use affects soil carbon storage, soil fertility, soil erosion, dust emission, and trace gas exchange. Land use practices in semiarid and arid regions can create large dust storms. Such storms transport dust and other aerosols from Asia across the Pacific to North America and from the Sahel to the Americas. These aerosols affect climate by changing the radiation balance of the atmosphere and by affecting cloud
development. Deposition of iron-laden dust particles acts as a micronutrient source that promotes marine primary production and thereby alters the global carbon cycle.

6 Future Directions

6.1 MODEL DEVELOPMENT

Recognizing that changes in land cover alter climate, the next generation of land models currently under development includes dynamic vegetation to allow interactive coupling of climate and vegetation (Foley et al., 1998, 2000). Studies with such coupled climate-vegetation models show that vegetation amplifies the climate response to changes in solar radiation or atmospheric CO$_2$ (de Noblet et al., 1996; Levis et al., 1999, 2000; de Noblet-Ducoudré et al., 2000; Doherty et al., 2000).

Land models are being developed to include the carbon cycle. The biogeochemical effects of land cover change may offset or accentuate the biogeophysical effects. For example, northward expansion of boreal forest into tundra decreases surface albedo and stores carbon. The climate cooling resulting from lower atmospheric CO$_2$ concentration may offset the warming from lower albedo (Betts, 2000). Claussen et al. (2001) contrasted the biogeophysical and biogeochemical effects of boreal and tropical deforestation. Boreal deforestation cools climate as a result of higher albedo and warms climate due to carbon loss to the atmosphere but not enough to compensate for the biogeophysical cooling. Tropical deforestation warms climate due to reduced evapotranspiration and releases carbon to the atmosphere resulting in warming that exceeds that from biogeophysical processes. Cox et al. (2000) also found that a dieback of tropical rainforests with a warmer, drier climate releases carbon that reinforces the warming.

Land models are being developed to better represent land use. Most models represent croplands as a single plant type with prescribed leaf area derived from satellite data. This fails to represent the variety of cropping practiced throughout the world. In the Great Plains, inclusion of interactive crops, especially the phenology of crop growth, improves simulated climate (Tsveltsinskaya et al., 2001a,b). Soil degradation alters infiltration, runoff, and water holding capacity and thereby affects climate. Urban land cover is excluded in global climate models, but is important at regional scales and will become more important as the world’s population congregates in megacities.

6.2 SURFACE DATASETS

Models such as BATS and SiB represent the land surface through a discrete number of pre-determined land cover types (Sellers et al., 1986; Dickinson et al., 1993). Parameters such as leaf area index, albedo, and surface roughness are determined according to land cover type. In the case of SiB, each grid cell is assigned to one of twelve land cover types, and look-up tables associate the cover type with a suite of parameter values. BATS represents the land surface with 16 cover types.

Originally, the land cover datasets used as boundary conditions in these models were derived from ground surveys, atlases, and other sources (e.g., Matthews, 1983; Olson et al., 1983). In the mid-1990s, global land cover classifications based on remote sensing data became available, first at a coarse resolution of 1 x 1 degree (DeFries and
Townshend, 1994) and later at higher resolutions of 8km (DeFries et al., 1998) and 1km (Friedl et al., 2002; Hansen et al., 2000; Loveland et al., 2000). These classifications label each pixel according to the unique spectral signature and phenological profile of each cover type. Additionally, statistical data fusion techniques combine satellite data with contemporary land cover surveys to better represent the extent and intensity of croplands and natural vegetation and to recreate historical land use (Ramanakutty and Foley, 1998; Cardille and Foley, 2002). Further refinements of land models also incorporated remotely sensed data directly. In SiB, the remotely sensed Normalized Difference Vegetation Index (NDVI) determines values for fraction of absorbed photosynthetically active vegetation, leaf area index, and roughness (Sellers et al., 1996a,b). The use of these remotely sensed data improves the depiction of landscape heterogeneity within cover types, but does not allow subgrid mixtures within grid cells.

Many of the early simulation experiments establishing the sensitivity of climate to land cover change are based on models that crudely represent the land surface with a discrete number of land cover types. These experiments alter the land cover type over grid cells that are large relative to realistic changes in land cover from human land use. For example, experiments replacing large areas of the Amazon Basin with grassland are important to establish the climate sensitivity to hypothetical land cover change (Dickinson and Henderson-Sellers, 1988; Nobre et al., 1991; Henderson-Sellers et al., 1993; Costa and Foley, 2000). However, homogeneous land cover alterations over grid cells that cover thousands of square kilometers are not realistic. Typically, land use alterations occur on spatial scales of 250m or less (Townshend and Justice, 1988).

As an alternative to characterizing the land surface with a discrete number of cover types, researchers have pursued approaches to map subpixel mixtures of vegetation at the global scale from remotely sensed data (DeFries et al., 1995, 1997, 2000a,b). “Continuous fields” describe the land surface in terms of the pixel’s fractional cover of plant functional types. Plant functional types allow increasingly sophisticated land surface models to assign parameters according to the vegetation’s physiological processes of carbon uptake and transpiration of water through stomata (Bonan et al., 2002). Traditional land cover types, on the other hand, most often contain mixtures of plant types with differing physiological properties, so that parameterization is not straightforward.

Continuous fields have several advantages. They overcome the artificial boundaries inherent in a traditional classification approach for more realistic depictions of the land surface. A classification approach, by definition, cannot capture mosaics and gradients that occur over much of the landscape (DeFries et al., 1995). The fractional coverage estimates vary across the landscape and consequently depict gradual transitions in vegetation type and subpixel mosaics within the grid cell size of land surface models. They also provide flexible characterization of the land surface as plant functional types that do not depend on a particular model’s land cover classification scheme. Because the continuous fields characterize each grid cell as the fractional coverage of plant functional types, the fields can be combined into any particular land cover classification scheme according to the cover type definitions. Alternatively, the fields can be directly used as boundary conditions in a land surface model (Bonan et al., 2002). Finally, continuous fields offer a means to incorporate realistic depictions of land use change in land surface models. Because human processes typically operate at spatial scales considerably smaller than a model grid cell, sub-pixel mixtures need to be incorporated in model simulation studies to realistically test the sensitivity to land use
change. Continuous fields, applied to a time-series of observations, can characterize land use change for input into these sensitivity studies.

Figure 2. Continuous fields derived from MODIS data at 500m resolution for 2000 (Hansen, unpublished).

To date, data from the AVHRR sensor have been used to derive “continuous fields” for several plant functional types, including woody vegetation, herbaceous vegetation, and bare ground (DeFries et al., 1999, 2000a). Woody vegetation has been further characterized by leaf type (needleleaf and broadleaf) and leaf longevity (evergreen and deciduous). Efforts are currently underway to derive the continuous fields globally from MODIS data at 500m resolution (Hansen et al., 2002b) (Figure 2). The continuous fields are derived by applying a regression tree, using training data from high resolution Landsat data and annual metrics describing vegetation phenology (e.g., annual maximum NDVI, minimum red reflectance, etc.) based on an annual time series of coarse resolution AVHRR or MODIS data (Hansen et al., 2002a,b). Applying the continuous fields method to multiple years depicts clearing and regrowth of forests, though this approach has only been possible to apply to the coarse resolution AVHRR 8km Global Area Coverage time series (DeFries et al., 2002b; Hansen and DeFries, submitted). Nevertheless, the ability to identify changes in forest cover through the continuous fields approach illustrates possibilities to assess climate feedbacks from realistic, sub-pixel characterizations of land use change.

Our ability to assess the effects of land use change on climate is improving as a result of advancements in three areas: 1) the next generation of land surface models with increasingly complex representation of vegetation processes; 2) abilities to derive continuous fields for the full suite of plant functional types represented in the models; and 3) availability of globally-comprehensive remotely sensed data from MODIS and other sensors that enable identification of land cover change over large areas.
7 References


LAND USE AND CLIMATE


Hansen, M., and DeFries, R. Detecting long term forest change using continuous fields of tree cover maps from 8km AVHRR data for the years 1982-1999. *Ecosystems*, submitted.


CHAPTER 18

URBANIZATION

CHRISTOPHER D. ELVIDGE¹, PAUL C. SUTTON², THOMAS W. WAGNER³, RHONDA RYZNER⁴, JAMES E. VOGELMANN⁵, SCOTT J. GOETZ⁶, ANDREW J. SMITH⁶, CLAIRE JANTZ⁶, KAREN C. SETO⁷, MARC L. IMHOFF⁸, Y.Q. WANG⁹, CRISTINA MILESI¹⁰, RAMAKRISHNA NEMANI¹⁰

¹NOAA National Geophysical Data Center, 325 Broadway, Boulder, Colorado 80303, USA. chris.elvidge@noaa.gov
²Department of Geography, University of Denver, Denver, Colorado USA
³China Data Center, University of Michigan, Ann Arbor, Michigan USA
⁴Tufts University, Boston, USA
⁵SAIC, USGS Eros Data Center, Sioux Falls, South Dakota USA
⁶Department of Geography, University of Maryland, College Park, Maryland USA
⁷Center for Environmental Science and Policy, Institute for International Studies, Stanford University, Stanford, CA 94305 USA
⁸NASA Goddard Space Flight Center, Greenbelt, Maryland USA
⁹Department of Natural Resources, University of Rhode Island, Kingston, RI 02881 USA
¹⁰University of Montana, Missoula, Montana USA

1 Introduction

Urban places may be broadly defined as the settlements where most people live and work. Human beings worldwide tend to cluster in spatially limited habitats occupying less than 5% of the world’s land area. Urbanization may be defined as those environment altering activities that create and maintain urban places. This includes the processes of construction, habitation, transportation, energy and water use, communication, industrialization, commercial and manufacturing services, plus civic activities associated with education and governance. The physical patterns of urban areas produce distinctive spatial and spectral signatures that are recorded by many types of remotely sensed data.

The seven thousand year old history of urbanization can be seen as a consequence of evolving technological capabilities to harness resources to support greater and greater human populations and enhanced occupational specialization and diversification. Today more than half of the world’s population lives in urban areas, with the most rapid increases occurring in the developing countries of Latin America, Asia, and Africa. In Europe, North America, and Japan 80% or more of the population already lives in urban areas.

Worldwide, the trend is for increasing numbers of people to concentrate in settlements and for the settlements to expand at their perimeters. Sprawl on the urban fringe and exurban development are two of the more conspicuous signs of urban change but structural change permeates urban areas through continuous redevelopment and the replacement of aging infrastructures with new constructions. Thus, urban areas are in a
constant state of redevelopment and flux that reflect both growing urban populations and the evolution of urbanizing technologies.

Different urban land uses (residential, commercial, industrial, recreational, etc.) typically have characteristic proportions of land covers and these proportions may vary with geography, climate, land availability, and social and economic custom. For example, most urban land uses include buildings, transportation infrastructure (e.g. streets and sidewalks), and open spaces but in different proportions. Commercial and industrial areas have significantly greater areas devoted to buildings and transportation (impervious surfaces) than residential areas. Residential land uses are often dominated by lawns, parkways or other types of cultivated open spaces. While the proportions of constructed (impervious) and vegetated surfaces vary, urban areas in the United States generally range from 30 to 70% constructed surfaces, with most of the rest being vegetated. In regions of the world with lower per capita incomes and/or land area limitations the percent cover of constructed surfaces in urban areas are higher.

The man-made construction of roads, airports, parking lots, and buildings are major disturbances to the natural landscape. This construction is different from other types of disturbances in that the possibility for ecological recovery and natural biological succession is arrested by the man-made materials that are resistant to decomposition and actively maintained. Many of the environmental changes associated with urban areas result from the use of impervious construction materials. By definition, impervious surfaces increase precipitation runoff and alter the hydrology of the local watersheds. These impervious surfaces replace vegetation, fragment habitats and alter the terrestrial water cycle. Likewise, impervious surfaces heat up very quickly from solar radiation and result in the locally elevated air temperatures of urban heat islands.

2 Urban Scales and Observables

In terms of scale there are two end-members in orbital remote sensing data. With satellite data having pixels 0.1 to 5 meters in size, it is possible to see important components of the urban landscape (roads, buildings, bridges, trees, lawns, etc.) and to construct maps of land cover and interpreted land use. It is even possible to produce three-dimensional representations of urban areas showing topography and individual buildings. However, the data acquisition and processing costs associated with the use of high resolution data restrict their practical use to relatively small areas.

At the coarser spatial resolutions of Landsat data (15 to 30 meters), it is possible to map larger land cover classes and interpret land uses. This is accomplished by a combining spatial and spectral differences between different land cover classes. With night time infrared bands (Landsat bands 5 and 7) it may be possible to observe the locations of high levels of energy combustion associated with commercial or industrial processes.

Coarse resolution data, with pixels on the order of 0.5 to 1 kilometer can also be used to identify urban features, although the level of detail is more generalized and, in some cases, reduced to a binary urban versus non-urban differentiation. Some examples include the identification of urban heat islands that contrast with the ambient temperatures of rural areas in thermal imagery, the spectral mapping of dense urban
structures using daytime visible and infrared data, and the detection of nocturnal lights using low light imaging data.

3 What is Urban?

The very concept of ‘urban’ as a landcover can be defined in a variety of ways. Housing density and population density thresholds are used by the U.S. Census Bureau as the basis for Urban Area (UA) designations. However, remotely sensed imagery is increasingly being used to map and monitor urbanization (Imhoff et al., 1997a and 1997b; Sutton 1997; Sutton et al., 2001; Lo 2002). Three benefits of remotely sensed imagery for urban applications are global availability, temporal resolution of measurement, and objectivity of measurement. These advantages are significant if one considers the inconsistent quality and availability of maps and census data around the world.

Night time satellite imagery collected by the DMSP-OLS has been used to demarcate urban area, estimate urban populations, and estimate urban population density (Ridd 1995). Daytime satellite imagery provided by Landsat has also been used to produce the USGS National Landcover Dataset which includes urban landcover at an even finer spatial resolution (Vogelmann et al., 2001). Remotely sensed imagery can also be used for performing urban ecosystem analysis based on hydrologic characteristics of urban land cover that can be identified by remotely sensed imagery (Ridd 1995).

Spatial measurement scale still represents a challenge with respect to documenting the processes of urbanization. Anderson Level II classifications of ‘Urban’ are: Low density residential, Medium density residential, High density residential, Commercial, Institutional, Industrial, Extractive, and Open Urban Land (Anderson et al., 1976). This level of classification is difficult to achieve with remotely sensed imagery alone, regardless of the spatial and spectral resolution. However, even coarse resolution night time lights imagery captures the ‘ex-urban’ very low density development that often exists outside many urban areas in what is often called the ‘urban-wildlands interface’.

4 Case Studies

4.1 FOREST LOSS AND FRAGMENTATION IN THE ATLANTA, GEORGIA REGION

Landsat data are extremely useful for assessing patterns and rates of land cover change related to urbanization. Many types of landscape changes, including urbanization, occur at relatively local scales, and the spatial resolutions of the various Landsat sensors (80 m for the earlier Multispectral Scanner (MSS) sensors and 30 m for the more advanced Thematic Mapper (TM) and Enhanced Thematic Mapper Plus (ETM+) sensors) are very appropriate for investigating such phenomena. However, perhaps even more importantly, Landsat data are available for much of the Earth’s land surface since the time of launch of Landsat 1 in 1972, making it possible to monitor landscape changes that have occurred over the past 31+ years.
Many researchers have successfully employed Landsat data for assessing urban changes (e.g. Royer et al., 1988; Sohl, 1999; Yang and Lo, 2002). Much less has been done with Landsat data to assess the impacts of urbanization. This case study demonstrates the use of Landsat data to assess one particular and deleterious side effect of urbanization prevalent in forested regions: forest fragmentation.

Forest fragmentation is a land cover conversion process in which larger forest tracts are dissected into smaller forest units. In some cases, deforestation in a large forest unit can result in many small units, whereas in other cases, the large forest unit still exists largely as a single unit, but the internal integrity and the boundaries are altered to become more “patchy” in structure. In either case, these types of fragmentation alter the spatial properties of the original forest, which in turn alters the ecological properties of the forest and the region’s biodiversity. Many organisms are well adapted to forest environments that are reasonably large and intact, and fragmentation can alter the forest enough such that these organisms can no longer survive in these areas (e.g. Gardner et al., 1993; Burke and Nol, 2000; Boulinier et al., 2001).

We used Landsat data from October 6, 1974 (MSS), October 3, 1987 (both MSS and TM), October 3, 1993 (TM), and September 28, 2000 (ETM+), over Atlanta, Georgia. These data sets represent near anniversary dates, thus ensuring greater levels of comparability among data sets and mitigating processing artifacts related to different phenological variables. The city of Atlanta has been one of the fastest growing metropolitan areas in the United States over the past 25 years, and was the site of an extensive Landsat-based monitoring investigation by Yang and Lo (2002). In that study, it was found that urban sprawl was accompanied by loss of forest. In the current study, the integrity of the region’s forests is the primary focus. In addition to the 1974, 1993 and 2000 single scene acquisition, both Landsat MSS and TM data were acquired for the same date in 1987, and the two data sources were compared to facilitate comparison and provide a “crosswalk” between the more coarse MSS data (1974) and the more advanced TM/ETM+ data.

The Landsat data sets were georeferenced to a common projection (Albers Equal Area) with root mean square errors generally in the order of 0.5 pixels or less. The data were then radiometrically normalized to the 2000 data set using pseudoinvariant objects (Schott et al., 1988). Forest versus nonforest classification data sets were then generated for each data set using a simple method of band thresholding described elsewhere (Vogelmann, 1995). For the most part, the same threshold values for defining forest versus nonforest were used for each data set, and thus the resultant classifications were very comparable among data sets for discriminating forested from nonforested regions.

A composite image showing a 50 by 50 km area including the city of Atlanta indicates that deforestation events have occurred throughout much of the region since 1974. Extensive patches of urbanization-related deforestation are especially prevalent to the southeast and northwest of Atlanta. Although this image is effective in portraying where deforestation events have occurred, it is difficult to assess ecological relevance without conducting additional analyses.

We developed a very simple and straightforward index of forest fragmentation that we refer to as the Forest Intactness Index (FI). The concept behind this index is that any pixel that is surrounded primarily by forested pixels represents a reasonably intact forest. A moving 7x7 filter window was used to identify intact forests. If 90% or
more of the pixels within the 7 x 7 window were classified as forest, the center pixel was coded as “intact.” Thus, FI images were generated for each of the five forest/nonforest data sets generated from the original Landsat data sets. It should be noted that many landscape indices have been developed, many of which are appropriate for assessing forest fragmentation (e.g. Wickham and Norton, 1994; Vogelmann 1995; O’Neill et al., 1996; Ritters et al., 2002). The FI described here was chosen because it is very basic and is quick to apply to the imagery. It is very likely that it is highly correlated with some of these other measures.

The FI was then summed for each of thirteen 10-km blocks that were placed along a west-east transect through Atlanta (the center of the city was in the center of the “middle” block of the transect) and converted to a per block percentage. Low percentage FI values for a given block imply low levels of forest intactness, whereas high percentage FI values for a block imply high levels of forest intactness. It was observed that for each block, the FI values for the 1987 data sets were consistently higher than for the 1987 TM data sets, which probably relates to the spatial resolution differences between MSS and TM sensors. In fact, this relationship was extremely consistent, and MSS-derived FI values regressed against the TM-derived FI values yielded an R² value of 0.995. The equation that describes this relationship was used to convert the 1974 MSS FI values to appropriate TM FI equivalents to enable better comparisons between MSS and TM/ETM+ epochs.

![Forest Intactness Along Transect Through Atlanta](image_url)

Figure 1. Transect through Atlanta, Georgia depicting changes in forest intactness (FI) both as a function of distance from the metropolitan center and as a function of time.

The results from this analysis are very effective in portraying both how the FI changes as a function of distance from Atlanta and as a function of time. Note that FI is very low for the block that includes the center of Atlanta in all four dates, but that FI increases as a function of distance away from the city center. In most of the cases, the 1974 data set has the highest levels of FI, implying that forest intactness was higher in 1974 than in subsequent dates. Most typically, the 2000 data set has the lowest FI
values. Both 1987 and 1993 FI values are generally intermediate (and are often indistinguishable from each other).

4.2 URBAN EXPANSION IN THE CHICAGO, ILLINOIS REGION

The Chicago metropolitan region experienced dramatic land cover change in the past decades. Redistribution of population and decentralization of metropolitan functions produced growth and development of outlying suburban areas. An important step towards a regional analysis is to understand the pattern and the quantitative results of urban land change. This study focused on the eight northeastern Illinois counties, which encompass the city of Chicago and its suburbs in the Cook, DuPage, Grundy, Kane, Kendall, Lake, McHenry, and Will counties. The study utilized three scenes of Landsat images acquired in 1972, 1985, and 1997, respectively, to provide a historical view of the urban areas.

Table 1. Land cover change in metropolitan Chicago region between 1972 and 1997.

<table>
<thead>
<tr>
<th></th>
<th>1972 (ha.)</th>
<th>1985 (ha.)</th>
<th>1997 (ha.)</th>
<th>1972–85 (ha.)</th>
<th>1985-97 (ha.)</th>
<th>1972–97 (ha.)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Urban Land</td>
<td>227,487</td>
<td>260,498</td>
<td>338,612</td>
<td>+33,011 (14.5%)</td>
<td>+78,114 (30%)</td>
<td>+11,1125 (48.85%)</td>
</tr>
<tr>
<td>Natural Area</td>
<td>227,047</td>
<td>209,796</td>
<td>179,061</td>
<td>-17,251 (7.6%)</td>
<td>-30,735 (14.65%)</td>
<td>-47,986 (21.13%)</td>
</tr>
<tr>
<td>Agriculture</td>
<td>599,462</td>
<td>468,804</td>
<td>375,537</td>
<td>-130,658 (21.8%)</td>
<td>-93,267 (19.89%)</td>
<td>-223,925 (37.35%)</td>
</tr>
</tbody>
</table>

The change detection result showed that urban land expansion dominated the regional pattern of land cover changes. Urban land includes all man-made features such as buildings, residential developments, cemeteries, roadways, landfills, quarries, and urban grasses. Between 1972 and 1985, urban land increased by 14.5%. Urban land increase and suburban sprawl accelerated to about 30% between 1985 and 1997. Within 25 years between 1972 and 1997, urban land increased about 49%. Most of the suburban land expansion occurred by consumption of agricultural land in the outer-ring of counties. Approximately 20% of agricultural land was converted into other land use in the time periods between 1972 and 1985, and again from 1985 to 1997. Natural areas that include forests, prairies, shrub lands, and wetlands declined 7.6% from 1972 to 1985, and 14.5% from 1985 to 1997. In total, 21% of natural areas were converted into other types of land covers in the 25 years (Table 1).

Data in Table 1 agreed well with the result derived by other institutions using different methodology but provided more details. A report by the Northeastern Illinois Planning Commission (NIPC) states that developed land in metropolitan Chicago increased 46% between 1970 and 1990, whereas its population grew by 4%. NIPC
projected that the population of the region will increase by 25% over the next 25 years (NIPC, 1998). Suburban sprawl and other urbanization processes will continue in the region. Historical remotely sensed data are critical in development of quantitative analysis models and in simulation of the effects of urban land cover change on regional natural and cultural systems (Wang and Moskovits, 2001). Based on the research finding from this study coupled with the population and employment projections made by the census and planning agencies, simulation of land cover change in selected areas to the year 2020 has been conducted (Wang and Zhang, 2001).

4.3 URBAN EXPANSION IN CHINA

The Pearl River Delta, situated in southern China, is one of the fastest growing regions in the country, and provides an excellent case study of the use of remote sensing to monitor urbanization. With disorienting speed, agrarian communities and vast tracts of agricultural land are being metamorphosed by a changed economy and enveloped by extended metropolitan regions. Prior to the economic and political reforms in the late 1970s, urban planning and urban growth were managed largely from the central government in Beijing. However, with the opening of the Chinese economy and the introduction of market-oriented policies, urbanization has been increasingly unstructured, unregulated, and chaotic. To understand the spatial and temporal dynamics of urbanization process, Landsat Thematic Mapper images were used to estimate urban land use change in the Pearl River Delta between 1988 and 1996.

Table 2. Area estimates of various land-uses and land-use change in the Pearl River area, China from 1988 to 1996.

<table>
<thead>
<tr>
<th>Class</th>
<th>km²</th>
<th>% of study area</th>
</tr>
</thead>
<tbody>
<tr>
<td>Water</td>
<td>2110</td>
<td>7.82</td>
</tr>
<tr>
<td>natural vegetation</td>
<td>11906</td>
<td>44.13</td>
</tr>
<tr>
<td>agriculture</td>
<td>10166</td>
<td>37.68</td>
</tr>
<tr>
<td>Urban</td>
<td>720</td>
<td>2.67</td>
</tr>
<tr>
<td>agriculture to water</td>
<td>16</td>
<td>0.06</td>
</tr>
<tr>
<td>natural vegetation to water</td>
<td>5</td>
<td>0.02</td>
</tr>
<tr>
<td>water to agriculture</td>
<td>151</td>
<td>0.56</td>
</tr>
<tr>
<td>natural vegetation to urban</td>
<td>529</td>
<td>1.94</td>
</tr>
<tr>
<td>agriculture to urban</td>
<td>1376</td>
<td>5.09</td>
</tr>
</tbody>
</table>

By applying a conceptual model of identifying land use change based on a nested hierarchy of associated land cover, a highly accurate urban land use change map was developed. Results indicate that between 1988 and 1996, the Pearl River Delta experienced a scale of urban land conversion unparalleled in the world; approximately
1905 km$^2$ of land was converted to urban uses during this period, an increase in 364% (Seto et al., 2002). From an estimated 720 km$^2$ of urban area in 1988, or 2.67% of the study area, the Delta’s urban land increased to over 2625 km$^2$ by 1996 (Table 2). Nearly 10% of the Delta is now urban. While roughly a quarter of the new urban areas were previously natural vegetation or water, most were converted from farmland, approximately 1376 km$^2$. Most of these agricultural areas were used previously for paddy rice, sugar cane, and lychee production. These remote sensing estimates of total agricultural land converted to urban uses are considerably greater than the 789 km$^2$ reported in statistical yearbooks (Seto et al., 2000). The central and provincial governments recognize the threat of declining agricultural land, and have supported initiatives to reclaim the Delta for agricultural purposes. Indeed, approximately 151 km$^2$ of the Delta’s water areas were converted into farmland. However, this increase in agricultural land offsets the loss in agricultural land by only 11%.

Historically, Pearl River Delta has been a rural region with a long agricultural tradition. Urbanization in the region is being driven by a myriad of compelling forces at multiple scales that are manifested in a number of distinct spatial configurations. The use of time series remote sensing data offers a unique opportunity to monitor the rapid urbanization process in one of the most highly dynamic regions in China.

4.4 THE GREENING OF DETROIT

American metropolises continue to experience dramatic changes as their core cities lose population and their economic activity shifts to sprawling suburban communities on their edges (Garreau 1992; Geddes 1997; Bogart 1998). Many of these metropolitan areas show rapid land cover changes, even as the growth of their populations slows or, in some cases, declines (Benefield & Chen, 1999). These changes reflect the dynamics of complex self-organizing systems, where population density and the intensity of human activities shifts in response to local economic conditions (Allen 1998; Bertuglia et al., 1998; Wilson 2002). The aggregate affect is patterns of changing landscapes that are observable from satellite altitudes.

Researchers at the University of Michigan used digital Landsat data of Wayne County, Michigan, to demonstrate the dynamic link between urban physical and social environments (Nystuen et al., 1966; Emmanuel 1997; Ryznar and Wagner, 2001). Wayne County is located in southeastern Michigan and wholly contains the 138 sq. mile City of Detroit, 10$^{th}$ most populous city in the United States (U.S. Census, 2000). Detroit is the first large U.S. city to lose almost half of its population over the past half century, even as the metropolitan area nearly doubled in size (SEMCOG 1999). Landsat MSS data from 10$^{th}$ May 1975 and 16$^{th}$ May 1992 were radiometrically corrected and processed to create a Tasseled Cap (TC) greenness features for Wayne County. Examination of greenness histograms allowed selection of thresholds above which the pixels were considered to be “green”.

The 1975-92 pixel-level vegetative greenness changes for the 615 sq. mile Wayne County and adjacent area of Canada. It also provides a comparison of Detroit’s vegetative change with its changes in income between 1970 and 1990. Statistical analysis showed the greening to be negatively correlated with population change ($r = -0.74$) and mean annual income changes ($r = -0.61$) but positively correlated with women/children ratios, a measure of household poverty ($r = +0.48$) (Ryznar, 1998). This inner city pattern of vegetative change coincides with neighborhoods of growing
poverty and structural abandonment – which contrasts to urban planning doctrines that actively promote new green space as a way to combat decay. A 1990 City survey showed that over 119,000 houses and other buildings had been torn or burned down since 1970, leaving behind vast grass or shrub-covered abandoned spaces in once thriving neighborhoods (Detroit, 1990).

In contrast to the above, increasing greenness on the urban fringe coincides with the conversion of agricultural land to urban development. Ground observations recorded extensive new residential subdivisions on previously cultivated land and the idling of cultivated land in anticipation of future urban development (SEMCOG, 2000). The result was the replacement of active agricultural land (bare in the spring) with both old-field vegetative growth and new perennial lawns, parks, and landscaping.

The simultaneous changes in vegetation in Detroit’s inner city and on its metropolitan fringe suggest connections between these two processes and provide graphic evidence of the demographic transitions that characterize mature cities and contribute to land cover/land use change.

4.5 IMPERVIOUS SURFACES AND SPRAWL DEVELOPMENT IN THE CHESAPEAKE BAY REGION

Rapid urban expansion, commonly known as urban sprawl, is often at the forefront of local political disputes. In many cases local governments make decisions on future growth without the benefit of reliable information on the full extent and environmental consequences of the current development. As a result, an ability to map the spatial extent of the built environment at high resolution has substantial relevance to local land use planning issues, with direct societal and economic relevance. Multitemporal Landsat Thematic Mapper (TM) and Enhanced Thematic Mapper (ETM+) data have been used successfully to track the progress of urban sprawl, providing critical input to the land use planning process. We report here on an approach to mapping impervious surface areas (ISA), including buildings, roads, parking lots) recently developed (Smith et al., 2003). The approach uses a regression tree algorithm that is trained with high resolution data sets (e.g., Ikonos, Quickbird, digital orthophotos and/or planimetric maps).

Subpixel Landsat estimation of ISA can be calibrated using high resolution reference data, such as aerial photography, high spatial resolution satellite data, or even planimetric charts. The algorithm searches for a dependent variable that, if used to split a population of pixels into two groups, explains the largest proportion of deviation of the independent variable (Brieman 1984). At each new split the tree is grown until it reaches terminal nodes, each representing a specific range of ISA identified with the training information. The regression tree outputs a continuous estimate of ISA between 0-100%) that can then be mapped over large areas using the Landsat data.

We used county-level planimetric data to train the ISA algorithm and then map, using year 2000 multitemporal Landsat imagery, the entire 168,000 km² Chesapeake Bay watershed. Whereas ISA was relatively easily detected with the Landsat imagery, careful iterative pruning of the regression tree was required for discrimination (and elimination) of areas with similar spectral properties, such as bare soil in agricultural fields. The resulting Landsat ISA map had an overall map accuracy of over 88%, as assessed with the Ikonos and DOQ images. There was some evidence
for systematic commission errors resulting from residual bare or plastic-covered agricultural fields, and beaches (Smith et al., 2003).

Application of the approach to additional years (1986, 1990, 1996) using just leaf-on and leaf-off TM imagery allowed us to produce a binary map of areas in the Baltimore – Washington DC region that had either greater or less than 10% ISA. This threshold was used to define a subpixel estimate of developed areas, but should not be confused with the total watershed value frequently used as an indicator of potential water quality impairment (Scheuler 1994). Accurate mapping of changes from vegetation to developed areas clearly delineated not only specific building projects, but also permitted us to record the time of development and calculate change rates through time. There was rapid development in the region throughout the observational period, with the highest rates of change occurring between 1986 and 1990. Our analyses suggest that new development in many areas has not been well planned, for example, there has been substantial land conversion greater than 10km from mass-transit stations. This characteristic of exurban sprawl typically induces greater automobile traffic, higher fuel consumption, inefficient land use, and additional stress on community infrastructure such as water, sewer, schools, etc. (Bockstael 1996).

To further investigate some of these sprawl impacts we are using of the time series of ISA maps, and related land cover type classifications, to inform spatial predictive models of urban change. Early results using the Sleuth cellular automata model (Clarke 1997) allowed us to produce maps of expected future change under various policy scenarios. Results of the simulations out to 2030 suggest rates of change under a current trends scenario may be up to ten times past rates, owing to the exponential nature of growth around existing developed areas (Jantz et al., 2003). We are also exploring the limitations of the cellular automata approach by simulating the economic bases of land use change decisions that account for parcel-level historical information, exogenous factors such as surrounding land use, and related variables that account for the land owner decision making process (Bockstael 1996). The power of these spatial predictive models is their utility to state and regional planning agencies tasked with targeting protective or restorative measures on valuable resource lands, particularly those identified as likely areas of future change, while providing incentives for development in areas targeted for focused growth or redevelopment.

4.6 URBANIZATION IMPACTS ON THE TERRESTRIAL CARBON CYCLE

Urbanization affects both the sink (carbon sequestration potential) as well as the sources (carbon emissions from industrial and transportation activities) in carbon cycling. Consequently, the impacts of urbanization are of increasing importance for the global carbon cycling. As cities expand, the paving of the surface for buildings, parking lots and transportation removes a good portion of the land formerly occupied by crops, grasslands or forests. Estimates of net primary productivity (NPP), the net amount of carbon fixed by plants, of the remaining urban vegetation can be compared to that of the surrounding land to quantify the impact of these land cover changes on the regional photosynthetic capacity.

A regional assessment of the net effect of urbanization on NPP was carried out for the southeastern United States (Milesi et al., 2003), which has undergone one of the highest rates of landscape change and urban sprawl in the country. The methodology used night time lights data from the DMSP/OLS for the years 1992/93 and 2000,
corroborated with census data, to map the recent growth of urban areas. NPP for the urban areas and the surrounding region was estimated from 1 km data from the MODIS sensor onboard the Terra EOS-AM1 satellite applying the MOD17 logic (Running et al., 2000).

Table 3. Estimates by states of the southeastern United States (US-SE) of the fraction of total land area developed between 1992/93-2000 and of the associated loss in annual NPP.

<table>
<thead>
<tr>
<th>State</th>
<th>% Area of New Development</th>
<th>Total loss in NPP (Tg C y(^{-1}))</th>
</tr>
</thead>
<tbody>
<tr>
<td>Alabama</td>
<td>1.3</td>
<td>0.38</td>
</tr>
<tr>
<td>Florida</td>
<td>2.5</td>
<td>0.55</td>
</tr>
<tr>
<td>Georgia</td>
<td>2.0</td>
<td>0.63</td>
</tr>
<tr>
<td>Mississippi</td>
<td>1.1</td>
<td>0.26</td>
</tr>
<tr>
<td>N. Carolina</td>
<td>2.4</td>
<td>0.54</td>
</tr>
<tr>
<td>S. Carolina</td>
<td>2.4</td>
<td>0.37</td>
</tr>
<tr>
<td>Tennessee</td>
<td>1.7</td>
<td>0.30</td>
</tr>
<tr>
<td>Southeast US</td>
<td>1.9</td>
<td>3.04</td>
</tr>
</tbody>
</table>

The impact of urbanization on regional vegetation productivity depends both on the type of land cover that is transformed and on the NPP retained by urban areas. In the southeastern United States, urban areas tend to expand at low density into the surrounding forests, and the developed lots often maintain a high tree canopy. In this case, the NPP maintained by the urban areas can be substantial. To verify this hypothesis and estimate the total impact of urbanization on the NPP of the southeastern United States, we compared the urban NPP with the NPP of the pre-existing land cover. The pre-urban land cover was inferred by overlaying the night time lights data for the year 2000 on the 1992 land cover information derived from the NLCD data set (Vogelmann et al., 2001). The analysis based on the night time data indicated that during the years 1992-2000 the urban developed land in the region increased by 1.9% (Table 3), with more than half of these new urban areas built at the expense of previously forested areas. Estimates derived from the MODIS data indicated that land cover changes due to urban development that took place during the analyzed period reduced annual NPP of the southeastern United States by 3.04 Tg (Teragrams, 10\(^{12}\) g) of carbon per year, amounting to 0.4 % of the total NPP in 1992 (Table 3). It is the high tree cover of the cities in this region (Dwyer et al., 2000) that probably constrained the loss in NPP. However, the significant population growth in the same years may have increased the carbon emissions substantially.

Recent studies have also shown that urban heat islands promote earlier plant growth in the spring, leading to longer growing seasons for urban vegetation compared to rural areas (White et al., 2001). Such growing season responses, however, depend on the regional climatic conditions and land use practices (Imhoff et al., 2000). A globally comprehensive analysis of the impacts of urbanization on carbon cycling is now
possible with the availability of consistent estimates of urban expansion and the photosynthetic activity from satellite data.

5 Conclusion

While urban areas occupy only a small fraction of the earth’s surface, urbanization has risen to become one of the driving forces altering the earth environment. Urban areas are the focal points for the consumption of food, water, and energy. They are likewise the focal points for both air and water pollution. Most of the agricultural, fisheries, and resource exploitation activities which constitute the balance of human impacts on the environment are driven by the consumption occurring in urban areas. The widespread use of impervious construction materials results in vastly increased surface runoff, which alters the stream flow and biodiversity. Urban areas tend to be built on flat, low lying areas, often replacing wetlands. While percentage of the earth’s surface covered by urban land cover is small, their influence is enormous.

Remote sensing provides one of the major sources of information regarding the spatial distribution of urban areas and their features. With high spatial resolution imagery (0.1 to 5 meters) it is possible to map land cover types with some detail and measure variables such as percent cover of impervious surface directly, but generally only for limited areas (tens to hundreds of square kilometres). With moderate resolution imagery (10-30 meters) it is possible to map major urban land cover types, land cover change, and build indexes for habitat fragmentation or percent cover of impervious surface for areas covering hundreds to thousands of square kilometres. While urban areas can be mapped and monitored with coarse resolution satellite imagery, the level of detail is limited. One advantage of using coarse resolution imagery is that it is possible to build global products on relatively short repeat cycles (e.g. annual updates).

In the coming decades urbanization will have a profound effect on the biological, chemical and climate systems of our planet, both at local and at global scales. Urban areas are the largest sources of anthropogenic greenhouse gas emissions, and major sources of aerosols and water pollution. Urban expansion results in losses of agricultural lands and the fragmentation of wildlife habitats. Urban areas will expand dramatically as human population numbers are expected to double in the next 60-70 years. The increased urban demands for energy, food and water will tax natural systems.

In no small measure, the activities concentrated in urban areas will determine the future habitability of our planet as a home for humankind and other species. There are legitimate questions regarding the sustainability of the current pattern of urbanization in the face of anticipated population and economic growth. Over time urban areas may evolve to become more compact, energy efficient and less polluting, exerting a smaller footprint on the environment. In the interim, remotely sensed data provide one of the best sources of information on how urban areas are changing through time and how they affect the environment.
6 References


CHAPTER 19

LAND USE AND FIRES


1University of Maryland, Department of Geography, 2181 LeFrak Hall, College Park, MD 20742 USA
2U.S. Geological Survey, Alaska Cooperative Fish and Wildlife Research Unit, University of Alaska Fairbanks, 214 Irving I, Fairbanks, AK 99775 USA
3Center for Global Change and Earth Observations 1405 S. Harrison Road, Room 101, East Lansing, MI 48823-5243 USA
4University of Maryland, Department of Geography, College Park, and NASA Goddard Space Flight Center, Code 922, Greenbelt, MD 20771 USA
5UFAC/WHRC Setor do Estudos do Uso da Terra e Mudanças Globais – SETEM Parque Zoobotanico Universidade Federal do Acre Rio Branco, Acre 69.915-900, Brasil
6USDA Forest Service, 1601 North Kent Street, Arlington, VA 22209 USA
7University of Zimbabwe, Institute of Environmental Studies, P.O. Box MP 167, Mount Pleasant, Harare, Zimbabwe
8Science Systems and Applications, Inc., NASA/GSFC, Code 923 Greenbelt, MD 20771 USA
9NOAA/NGDC, 325 Broadway Boulder, CO 80303 USA
10Canadian Forest Service, Great Lakes Forestry Centre, 1219 Queen Street East, Sault Ste. Marie, Ontario, P6A 2E5 Canada
11University of Alaska Fairbanks, School of Agriculture and Land Resource Management, P.O. Box 757140 Fairbanks, AK 99775 USA

1 Introduction

Research on fire is often of an applied nature, addressing questions of how to manage landscapes for fire, how to determine fire danger, how to model fire behavior, fire impacts and post-fire succession (Martell 2001; Chuvieco 2003). This in part reflects the desire of the funding agencies to maximize the benefits from the large amounts of public money spent each year on fire management. There is an increasing body of fire research in the area of global change, for example studying the potential impacts of climate change altering fire regimes, the impact on the atmosphere in terms of emissions, radiative forcing and chemical composition and feedbacks to the surface (Scholes et al., 1996; Stocks et al., 1998; Govaerts et al., 2002).

Fire is a global phenomenon and is an important agent of global environmental change. Fire has played an important role in shaping the current distribution of global vegetation and the paleo-record shows a close link between fire and early human life (Pyne 1995). Today fire remains an indicator of land cover change and is in many land use systems the proximate cause of land cover change (Janetos and Justice, 2000). The causes of fire are many and varied. In many parts of the world, fire is used as a land management tool for land clearing, pest control and removing crop residue. Lightning fires predominate in less
populated regions such as the boreal forests, while man-made fires associated with land clearing dominate the humid tropical forests and woodlands. In savanna ecosystems there is a mix of both anthropogenic and lightning fires. Humans have developed different fire management strategies for different ecosystems, often altering the natural fire regime.

Recent years of climatic extremes have resulted in catastrophic fires and have raised public awareness of the relationship between fires, human populations and climate variability (Dennis 2003). Although large areas of forest and grasslands are affected by wildland fire each year, increasing human population and demand for land means that more fire-prone areas are becoming inhabited and with this recognized risk, more land is subject to fire management. Similarly, traditional uses of fire for land management may no longer be tenable with increasing population and decreasing availability of land and commercial logging (Cochrane 2001a). In Siberia institutional changes are leading to changes in the extent of fire management and resulting fire regimes (Pyne 1995). In Indonesia extensive logging and traditional slash and burn farming, combined with extreme droughts are resulting in fires in rain forest areas of an unprecedented extent (Schweithelm 1999; Siegert et al., 2001). In the U.S., urban encroachment on wildland areas and increases in burned areas and in severity of fires in the end of the 20th century, particularly in forests dominated by historically short-return interval low-intensity fire regimes, have led to changes in fire management policies, with an increased emphasis on fuel management and use of prescribed fire. Implementation and monitoring of the effects of these policies will require vast improvements in spatially based data products and increased integration of remote sensing-derived information into planning and monitoring (Conard et. al, 2001). In the Mediterranean, where movement of people from the countryside to the city results in the abandonment of arable land, the increase in woodlands and associated fuel is changing the fire regime (Alexandrian and Esnault, 1999). Land use change is playing an increasing role in the global change research agenda and fire, as a land use tool and a major agent of land cover change over large areas of the World, is an important topic for research (Gutman et al., this volume).

Management of landscapes to minimize damaging fires is an extremely controversial and important policy issue and one that needs to be developed in the context of broader land use policy making (FAO 1999). Policies need to be developed which are appropriate for a given landscape and its use and which minimize the adverse effects of uncontrolled fires. Fire management policy is complex, and there are often conflicting environmental considerations, for example balancing the need for increased prescribed burning to better management of fire-dependent ecosystems, while minimizing loss of biodiversity or atmospheric pollution to protect human health. The study of fire often falls across disciplines and requires an integrated approach to research.

Prescribed burning also plays a role in national accounting of greenhouse gas emissions and there is a body of research developing to quantify fire emissions (Andreae and Merlet, 2001). While accurate greenhouse gas accounting is an element of emerging international climate change policies, trans-boundary pollution of smoke has elevated the fire policy to a different international forum (WMO 1998; Wotawa and Trainer, 2000).

In this chapter we examine fire as a component of the land use and land cover change research, the satellite systems at our disposal to study fire and summarize case studies on fire and land use, from three different regions.
2 Fire, Land Use and the Carbon Cycle

Fire has the potential to influence the atmospheric concentration of CO₂ by releasing carbon from terrestrial ecosystems to the atmosphere. Both anthropogenic and non-anthropogenic fire activity influence the carbon cycle at intra-annual, interannual, and longer time scales. Fires in Africa have a measurable effect on the seasonal cycle of atmospheric CO₂ recorded at Ascension Island in the Atlantic (Wittenberg et al., 1998). Similarly, fire activity in Siberia may play a role in the seasonal cycle of atmospheric CO₂ in high latitudes (Kasischke and Bruhwiler, 2002). It has only recently been elucidated that fire has enough interannual variability that it may play a role in the interannual variability in the growth rate of atmospheric CO₂ (Langenfelds et al., 2002). This variability is likely associated with the effects of climate on fire, as El Niño increases fire activity in both the tropical and the boreal zones to release terrestrial carbon to the atmosphere (Hess et al., 2001; Langenfelds et al., 2002; Page et al., 2002; Duffy et al., submitted). The increase in tropical fire activity during El Niño is primarily caused by lower precipitation, while the increase in the boreal zone appears to be driven by higher temperatures during the summer months (Hess et al., 2001; Duffy et al., submitted).

To understand how variability in fire activity influences longer term carbon storage of the terrestrial ecosystems, it is useful to examine the effects of forest fire on carbon storage at both the stand and regional levels. Stand-level disturbances, like fire, are generally characterized by a period of ecosystem carbon loss, during which production is less than decomposition after initial losses associated with fire emissions (Zhuang et al., 2002). The period of carbon loss is then followed by a period of ecosystem carbon gain once production exceeds decomposition, and has the potential to go to steady state as production and decomposition become equal (Zhuang et al., 2002). If regional fire regimes are at steady state for a substantial period of time, ecosystem carbon storage will not change because areas of carbon loss are offset by areas of carbon gain. However, an increase or decrease in fire frequency or severity will generally cause losses or gains, respectively, in carbon from terrestrial ecosystem to the atmosphere. Fire has clearly played a substantial role in changing long-term carbon storage of terrestrial ecosystems. In the conversion of forests to agriculture, fire is the proximate agent that removes carbon from wood biomass to the atmosphere. Between 1920 and 1992, it is estimated that the net effects of cropland establishment and abandonment released between 55 and 90 Pg C to the atmosphere (McGuire et al., 2001). Global land use is estimated to have resulted in the net release of around 2 Pg C of carbon annually to the atmosphere in the 1980s, primarily because of tropical deforestation (Houghton et al., 1999). Changes in non-anthropogenic fire regimes are also estimated to have substantial effects on carbon storage. Human fire suppression is estimated to be responsible for substantial carbon storage in the United States during the 20th Century as fuels have built up in fire-protected forests (Pacala et al., 2001). In contrast, an increase in fires in northwest North America in the 1970s and 1980s is estimated to have released substantial amounts of carbon to the atmosphere (Kurz and Apps, 1999).

Of potential responses to global change, fire is the disturbance agent that has the greatest potential in releasing large amounts of carbon from terrestrial ecosystems to the atmosphere in a short amount of time. If there is a link among increases in atmospheric CO₂, climate change, and increases in global fire activity, then there is the potential for a positive feedback loop in which fire plays an important role in climate change. Thus, it is important to understand the spatial and
temporal dynamics of regional fire regimes and the degrees to which human activities and climate influence fire regimes.

3 Fire and the NASA Land Cover and Land Use (LCLUC) program

Spaceborne sensors provide a unique perspective with which to study and understand the global distribution and characteristics of fire. NASA’s program of technology development, experimental satellite observing systems, scientific research and development, information systems development and earth science applications provide the means for an improved understanding of fire pattern and process. They also allow an end to end development and testing of global fire monitoring capabilities and their insertion into the operational domain. Through a program of competed research, the NASA LCLUC program is supporting different aspects of remote sensing oriented fire research (Gutman et al., this volume). Three regional fire studies are presented in Section 5.

Until quite recently, fire has not been considered an environmental record by the operational space agencies and the existing and planned operational satellites have limited fire monitoring capabilities (Townshend and Justice, 2002). Much of the development of fire remote sensing still remains in the research domain. The requirements for improved remote sensing of fires have been articulated by the fire management community (CEOS 2000). The challenge now is to transition the technology developed in the research domain into the operational domain.

4 Fire Observations and Monitoring Products

Comprehensive monitoring of fire involves observation of processes leading to fires, assessment of fire risk, detection of fire, mapping of fire affected areas, assessment of land cover disturbance and post-fire recovery processes, and estimation of atmospheric emissions. Several countries with significant wildfire activity have developed ground- and air-based monitoring networks. Satellite remote sensing provides opportunities for continuous, large-scale monitoring, which may overcome logistical and financial constraints of ground- and air-based observations.

Remote sensing instruments on polar orbiting and geostationary satellites allow fire observations at a broad range of spatial and temporal scales. Coarse spatial resolution (1-4 km) sensors on geostationary satellites provide hemispheric observations every 15-30 minutes. Sensors with broad swaths on polar orbiting satellites provide coarse to moderate spatial resolution observations (0.25-1 km) on a near daily global basis. High spatial resolution (<10 m – 100 m) narrow-swath sensors have longer revisit intervals and are needed for local scale applications.

Active fire mapping algorithms have been developed to take advantage of the elevated radiance signal from hot fires. Several sensors carry bands in the mid-infrared where the emitted signal from fires is usually higher than that from the surrounding surface. The Advanced Very High Resolution Radiometer (AVHRR) on the polar orbiting NOAA (National Oceanic and Atmospheric Administration) operational satellite series provide more than 20 years of global daily observations. A number of AVHRR-based fire monitoring systems have been developed by international initiatives such as the World Fire Web (Grégoire et al., 2001) and by national or regional agencies such as the Canadian Fire M3 system (Li et al., 2000) and the Russian Space Monitoring Information Support (Bartalev et al., 2001). The
Potential value of the AVHRR time series for reconstructing the history of biomass burning has long been recognized and supported by the NASA LCLUC program (Csiszar et al., 2003). Instruments on the NOAA Geostationary Operational Environmental Satellite (GOES) series have also been providing useful information since 1983. The GOES Wildfire Automated Biomass Burning Algorithm (ABBA) is providing half-hourly fire observations for the Western Hemisphere (Prins et al., 1998). An active fire monitoring system has been running using the nighttime visible fire signal measured by the Operational Linescan System (OLS) on board the Defense Meteorological Satellite Program (DMSP) series (Elvidge et al., 2001). Monitoring systems based on instruments on other environmental satellites have been developed. The mid-infrared channels of the European Space Agency Along-Track Scanning Radiometer (ATSR) and Advanced ATSR series have been used to create the World Fire Atlas (Arino et al., 2001). The Visible and Infrared Scanner (VIRS) on board the NASA Tropical Rainfall Measuring Mission (TRMM) satellites provide fire observations and, like GOES, allows sampling of the diurnal fire cycle (Giglio et al., in press). None of the above sensing systems were designed for active fire detection. A major step forward has been the successful launch in 1999 and 2002 of the Moderate Resolution Imaging Spectroradiometer (MODIS) on board the morning descending Terra and afternoon ascending Aqua polar orbiting NASA satellites. The MODIS sensor design includes bands specifically selected for fire and cloud detection and allows the retrieval of sub-pixel fire area and temperature (Kaufman et al., 1998). MODIS is currently the only system providing global daytime monitoring when the activities of active fires are the greatest (Justice et al., 2002; Figure 1).

![MODIS Land Rapid Response Fire Detections](image)

Figure 1. Monthly global distributions of active fires from Terra/MODIS. (Produced by MODIS Land Rapid Response System; see CD for color image)

Satellite mapping of fire-affected areas, also known as burned areas, has less heritage than active fire detection. This is despite the recognition that cumulative active fire detections may not reliably map the fire-affected area. Algorithms that use multi-temporal satellite data to take advantage of the temporal persistency of fire effects have recently received considerable attention for regional to continental scale mapping. These methodologies have relied on classification or thresholding techniques to label remotely sensed observations as burned or unburned. Traditional algorithms include the detection of changes in spectral indices such as the Normalized Difference Vegetation Index (NDVI) (Kasischke and French, 1995). Systematic AVHRR-based burned area mapping systems have
been developed that incorporate active fire detection data in boreal ecosystems (Sukhinin 1999; Fraser et al., 2000). In savanna and temperate ecosystems the change in reflectance caused by fire may be smaller than in boreal systems, as less fuel is burned, and the change is generally less persistent. In southern Africa MODIS observations have shown that the magnitude of the variations in reflectance due to angular sampling may be greater than the reflectance change induced by biomass burning (Roy et al., 2002a). Similar observations were made in northern Australia using SPOT-VEGETATION data (Stroppiana et al., 2002). Roy et al. (2002a) developed a bi-directional reflectance model-based expectation approach to map the 500 m location and approximate day of burning using daily MODIS observations. Recently, global burned area products have been created from ATSR (GLOBSAR; Simon 2002) and SPOT-VEGTATION (GBA; Grégoire et al., 2003) time series data, although these products have not been validated rigorously.

Fire risk assessment is based on the evaluation of surface and atmospheric conditions for fuel flammability (Chuvieco et al., 2002). Vegetation conditions are correlated with remotely sensed spectral indices, such as the NDVI, and with surface temperature estimates (Zarco-Tejada et al., 2003). Atmospheric conditions are either derived from analysis or forecast fields of numerical weather prediction models or from vertical profiles derived from satellite-based sounding instruments. Monitoring systems are emerging around the globe (Lee et al., 2002; Carlson and Burgan, 2003). Post-fire assessment of burn severity, re-growth, land cover disturbance and emissions are primarily in the research domain, with several projects supported by LCLUC (Conard et al., 2002).

As the importance of fire monitoring has gained scientific and political recognition, new sensors are being designed with improved fire mapping capabilities. The SEVIRI instrument on the Meteosat Second Generation (MSG) satellite, launched in 2002 provides an improved capability for fire monitoring in Africa (Schmetz et al., 2002), the continent with the most extensive biomass burning. The Visible and Infrared Radiation Sensor (VIIRS) on board the next generation National Polar Orbiting Environmental Satellite System (NPOESS) will be the first sensor on a polar orbiting operational satellite with specific fire detection features and superior geometric and radiometric characteristics than AVHRR (Townshend and Justice, 2002). Small satellite technologies are being prototyped by the German Aerospace Center’s (DLR) BIRD (Bi-spectral Infrared Detection) mission (Briess et al., 2002) and the FUEGO program led by the Aerospace Engineering and Services (INSA, Spain) towards the creation of a constellation of small polar orbiting fire satellites.

Recently there has been a trend toward integration of information provided by different sensors. Active fire products from various sensors may be viewed simultaneously using Internet and Geographical Information System technology. For example, the US NOAA Hazard Mapping System displays data from AVHRR, GOES imager, OLS and MODIS (McNamara et al., 2002), the South American PROARCO system displays data from AVHRR and MODIS (Justino and Andrade, 2000). The MODIS Rapid Response System displays fire data from sensors on Aqua and Terra (Justice et al., 2002). Little work has been undertaken to develop multi-sensor fire detection algorithms, or to assess the quality and accuracy of data derived from different systems for their synergistic use.

Validation of satellite-derived fire products is an essential component of the monitoring process (Morisette et al., 2002) and should be underpinned by routine quality assessment to ensure generation of consistent long term products (Roy et al., 2002b). The CEOS Working Group on Calibration and Validation Land
Product Validation Subgroup, in collaboration with GOFC/GOLD-Fire, is promoting the creation of a network of fire validation core sites with strong participation by partners in the regional GOFC/GOLD networks. In addition, partnerships with operational agencies have been formed for the exchange of satellite fire products to support fire management decisions, and ground- and air-based fire observations for continuous validation (Justice et al., 2003). Efforts are being made towards the development of integrated systems where ground-, air-, and space-based fire observations are merged in an optimum way to produce synergistic fire products.

A major challenge to the space agencies and international fire community is to transition the existing demonstration projects into fully operational products to ensure long-term, continuous fire observations over the globe with known accuracy. Within this process, efforts need to be made to extend the suite of “traditional” active fire and burned area products with more refined fire characterization.

5 Regional Assessments of Land Use and Fire

5.1 FIRE AND LAND USE IN THE HUMID TROPICS

Fire and land use are integrally linked in the tropics. For thousands of years fire has been an important tool for clearing forests and reducing their biomass into nutrient rich ash for crops. Tropical landscapes have become a dynamic collection of intact, cleared, degraded, and regenerating forests. In recent years, uncontrolled fires caused by current land use practices have become a great concern due to the vast area affected. These fires have damaged millions of hectares of tropical evergreen forest (Cochrane et al., 1999; Siegert et al., 2001; Page et al., 2002) and damaged the health of millions of people (UNEP 2002a).

Any response to forest fires in the tropics is premised on being able to locate these fires while they are occurring. While this is simple in theory (i.e., look for the smoke), both the culture of fire use in the tropics and the limitations of infrastructure and satellite technology make locating forest fires challenging (UNEP 2002a). In addition to clearing new lands, fire is used to remove regrowth every 2-3 years (Fearnside 1990; Kauffman et al., 1998). This signifies that, regardless of the presence or absence of forest fires, there will be tremendous numbers of fires across the landscape each year. As a result, the majority of fires detected each year is neither deforestation nor forest fires, but simply maintenance fires on previously cleared lands (Miranda and John, 2000). The salient point being that, not only do forest fires have to be detected, they must also be distinguished from thousands of intentional/beneficial fires that are set each year.

Land cover change exacerbates the fire problem in the tropics by increasing the prevalence of flammable ecosystems (e.g. grasslands) and the connectivity that exists between them (Figure 2). In the past, agricultural plots and pastures existed as islands of easily flammable vegetation within an area of largely fire immune forest. However, as a region develops, its forest remnants become increasingly fragmented and surrounded by easily flammable vegetation. Connectivity of these flammable ecosystems results in a greater chance of each fire escaping into neighboring pastures, directly increasing the total area exposed to fire.

Tropical forest fires are largely anthropogenic in nature and highly edge related (Cochrane 2001a). This edge effect can extend for kilometers into the forest, degrading even large tracts (Cochrane and Laurance 2002). Fires in tropical
rainforests range from easily extinguished litter fires to nearly impossible to extinguish ground fires. Once burned, a forest becomes degraded and prone to recurrent fires (Cochrane and Schulze 1999). In recurrent wildfires, average fire intensities are 10 times greater (307 versus 30 kW m\(^{-1}\)) and can spread twice as fast (0.52 versus 0.25 m min\(^{-1}\)) (Cochrane et al. 1999). While some species can survive the lower intensity fires, a positive feedback can become established wherein each successive fire becomes more intense, killing more of the remaining forest. This process can continue, with subsequent fires worsening until complete deforestation occurs (Cochrane et al. 1999).

![Diagram of interrelationships between tropical land cover changes and forest fires](image)

Figure 2. Schematic diagram of the interrelationships between tropical land cover changes and forest fires (Cochrane 2003). Arrows indicate forcing of each node upon others in the system. Gray arrows directly affect forest fire occurrence, black arrows indirectly influence forest fire occurrence.

- a. Road building results in forest access that is strongly associated with deforestation (Laurance et al., 2001).
- b. Deforestation fragments the remaining forests creating increasing amounts of edge (Skole and Tucker, 1993; Laurance et al., 1997).
- c. Road building and paving provide greater access to the forest to loggers and directly affect the transportation costs and area of economic accessibility (Veríssimo et al., 2002).
- d. Logging directly results in limited amounts of deforestation for roads and log landings. Post-logging colonization frequently leads to settlement and deforestation (Veríssimo et al., 1995).
- e. Forest edges are subject to biomass collapse and microclimate changes (Laurance et al., 1997) that make them very susceptible to frequent fires (Cochrane 2001a).
- f. Repeated forest fires, especially in previously logged forests, can lead to unintentional deforestation. Accidental deforestation can cause half of the total deforestation in some regions (Cochrane et al., 1999).
- g. Both deforestation fires and pasture/land maintenance fires result in many accidental forest fires (Cochrane 2001b) that predominate along forest edges (Cochrane 2001a).
- h. Logging degrades the forest and can lead to increased susceptibility to fire (Uhl and Buschbacher, 1985; Holdsworth and Uhl, 1997). This can lead to extensive fires even several years after the logging operations (Siegert et al., 2001).
- i. Forest fires can create a positive feedback cycle where recurrent fires become more likely and more severe with each occurrence (Cochrane et al., 1999).
Previous forest fires and selective logging both intensify the connectivity of flammable ecosystems on the landscape. Intact forests are still resistant to fire encroachment even after more than a month without rain, but selectively logged or burned forests may become flammable in as few as 6-15 days without rain (Uhl and Kauffman 1990, Holdsworth and Uhl 1997, Cochrane and Schulze 1999). Like fire, logging opens the canopy and allows the forest to rapidly lose moisture and dry out. These forests, with their heavy fuel loads and diminished canopies, provide the flammable vegetation that can link the region’s cleared lands and exposes more forest to potential fire events (Figure 3).

If fire incidence in many regions of the tropics stays at current levels or increases, many forests will be replaced by more fire tolerant vegetation over the coming decades. Although post-fire regeneration of trees can be robust (Uhl and Buschbacher, 1985; Kauffman 1991), frequent fires can prevent these trees from reaching reproductive ages (Cochrane and Schulze, 1999). The shift from a fire regime of little or no fire, characteristic of evergreen tropical forests, to one of frequent fire is consistent with that found in scrub and savanna (Hammond and ter Steege, 1998). In the more seasonal tropics, the destruction of the current forest cover is likely to be irreversible under current climate conditions (Mueller-Dombois, 1981; Shukla et al., 1990).

Figure 3. Positive and negative feedbacks controlling fire processes in tropical forests (Cochrane 2003). Positioning of processes or activities indicates whether they occur within or outside of the forest (green shading). Items bordered in red control fire occurrence or behavior, while items in brown modify the potential fire environment. Green indicates ecosystem processes that act in opposition to fire, specifically regrowth, canopy closure and decay of fuels. The management efforts and interventions box indicates how and where human actions can diminish tropical forest fires. Climate change encompasses the effects of
increased CO$_2$, land cover change, and aerosol loading the net effect of which is regional drying in the tropics (Ramanathan et al., 2001; DeFries et al., 2002; Kaufman et al., 2002). Changes in the amount or intensity of any of the processes or activities will change the strength of their forcing functions (arrows) but not their direction. For example, increasing education of rural people in better fire management techniques causes a reduction of fire ignitions entering the forest which, in turn, reduces the level of interaction with forest fuels that leads to increased fire severity. Fire ignitions, however, are always positively related with increasing fires. (See CD for color figure.)

Two major environmental changes that will influence regional climate change in the tropics are increasing levels of atmospheric CO$_2$ and continued deforestation. Increasing CO$_2$ levels tend to raise temperatures and precipitation levels while deforestation depresses overall precipitation levels in global climate models (Zhang et al., 2001). The net effect across the tropics will be a combination of these two disturbances. Model predictions of Amazonian precipitation for a doubling of atmospheric CO$_2$ show an increase of 0.28 mm/day. However, deforestation reduces average precipitation levels by 0.73 mm/day resulting in a net drying (Costa and Foley, 2000). Results are similar for all of the World’s tropical regions with deforestation-driven reductions in carbon assimilation yielding reduced evapotranspiration and hence a relative increase in the amount of sensible versus latent heat flux. The net effect is increased surface temperatures and drier hydrological conditions (DeFries et al., 2002) that could worsen future fire problems.

The crux of the fire problem in tropical rainforests is not so much the introduction of fire into these ecosystems but the frequency with which they are being burned. Landscape fragmentation and land cover change interact synergistically to expose more of the forest to fire and consequently raise the risk of unintended fires occurring across the entire landscape. As selective logging and periodic forest fires occur, the landscape matrix becomes increasingly porous, allowing fire contagion to easily spread across the landscape. With millions of hectares of fire-damaged forests spanning the tropics, studies of natural and managed ecosystem recovery are needed. In particular, rates at which these damaged rainforests recover their near-fire-immune status need to be determined. Potential intervention or management strategies to enhance ecosystem recovery processes should also be assessed (Cochrane 2003).

In order to move beyond simple charting of the frequency and timing of burning at the landscape level, it will be necessary to relate fire detection information and burn scar maps to up-to-date land cover maps. The ecological importance of fire can only be assessed through knowledge of what is burning, how often and when. The need for high sampling frequency using high-resolution imagery currently makes the mapping of forest fires in the tropics expensive and labor intensive (UNEP 2002a). Wherever possible, fire detection and burn scar mapping should be combined with ongoing land use and land cover mapping efforts in order to maximize the efficiency of resource use and the synergy of information that can be used to assess and interpret the causes and effects of fires across the tropics. Through such synergy of information, it will be possible to provide critical information such as when, where and what is burning so that effective fire management actions can be taken. This knowledge will also enhance the decision-making capacity of both policy makers and land managers who need landscape level understanding of the interaction between various land uses in order to promulgate effective land use strategies.
5.2 FIRE AND LAND USE IN AFRICAN SAVANNAS

Fire is prevalent throughout Africa. Across most of southern Africa the distribution of savannas – including arid shrublands, lightly wooded grasslands, deciduous woodlands and dry forests – coincides with climates characterized by a single annual rainy season and a long dry season (Scholes 1997; van Wilgen and Scholes, 1997). Desiccation and senescence of the vegetation during the dry season, from about May to October, produce ideal conditions for wild fires that, in turn, contribute to shaping and maintaining savanna structure. The savannas of Africa are thought to experience the most extensive biomass burning in the world (Dwyer et al., 2000; Scholes and Andreea, 2000; Andreea 1992), contributing, for example, up to 24% of all CO\textsubscript{2} derived from biomass burning worldwide (Andreea 1997).

Most fires in southern Africa are ignited by people, mainly for land management purposes. Others are caused by lightning associated with early wet-season convective thunderstorms. The use of fire is integral to many African agricultural practices, particularly in clearing land for cultivation. Fire is also used to control bush encroachment, provide a green flush to attract game and improve grazing for livestock, drive animals during hunting, construct fire breaks and for other security reasons (Frost 1999). Fires started by people early in the dry season are likely to pre-empt later-occurring lightning fires from igniting or spreading (Frost 1999).

Fuel loads in savannas are controlled primarily by annual precipitation and soil fertility. In wet regions, between 1,000-1,200 mm rain per year, there is sufficient rainfall to support closed-canopy woodlands and forests, which reduces grass production. Prolonged moist conditions and heterogeneous vegetation structure constrain the spread of fires even if fuel loads are high (van Wilgen and Scholes, 1997). Fires are larger and more contiguous in the grasslands and open savanna woodlands at intermediate rainfall amounts (about 550–750 mm per year) where there is sufficient grass production to produce hot fires that can damage trees, but not enough rainfall to allow rapid regeneration and closure of tree canopies, which would otherwise restrict grass production (Scholes et al., 2002). Below about 550 mm annual rainfall, where the vegetation is predominantly shrubby grassland, and grass production is strongly linked to annual rainfall, fire is intermittent and generally follows periods of well-above average rainfall. The availability of fuel is reduced by herbivory (e.g., domestic livestock, wild herbivores, and insects) and by people (e.g., fuel collected for domestic energy) (Desanker et al., 1997; van Wilgen and Scholes, 1997; Frost 1999). Fire is less frequent on fertile grasslands and in arid savannas where the consumption of palatable vegetation by herbivores leaves little to burn (Bond 1997; van Wilgen and Scholes, 1997). Scholes et al. (1996) suggested that infertile savannas and grasslands burn about once every 3, 2, and 1 years if they receive 600 mm, 900 mm and 1200 mm mean annual precipitation respectively, and that fertile equivalents burn approximately once every 5, 3, and 1 years.

The influence of fire on the vegetation dynamics of Southern Africa has long been recognized. While fire undoubtedly enhances the productivity and quality of some resources, notably grasses used for grazing, it diminishes others. These include timber and other wood products, grass used for thatching, and habitat for animals hunted by people. Exclusion of fire, either deliberately through fire control or inadvertently through reduced fuel loads in heavily grazed areas, can also have adverse effects, most notably bush encroachment, leading to a reduction in the grazing capacity of the land. Legislation in southern Africa requires people wishing
to set fires for management purposes to notify their neighbors and the police of their intentions, though this is seldom done. Fire management policies that include prescribed burning and the suppression of wildfires are followed where resources are available, for example, in privately owned wildlife reserves, commercial farms, along certain urban-forest interfaces, and in some national parks and conservation areas. With greater emphasis on ecosystem management, policies in conservation areas now give greater recognition to the importance of fire in maintaining biodiversity, and acknowledge also the inevitability of anthropogenic fires (due to accidental and arson fires, and wildfires spreading from surrounding land). Outside conservation areas, prescribed burning in the early dry season, to reduce the possibility of intense wildfires occurring later, provides a cost effective method for community fire management. On private land and national reserves, fuel loads may be reduced by practices that include lease-grazing and the collection of dead wood and grasses. How fire regimes will change as population size, distribution and land uses change is unclear. Changes in land-use intensity and land-management practices are likely to affect fire regimes directly, through changes in the ways fire is used, and indirectly, by modifying fuel loads and environmental conditions (Frost 1999). Certainly, restricted economic opportunities and lack of security of tenure will continue to force rural people to depend on low-input subsistence agriculture, thereby maintaining an ongoing need to open up new land rather than use existing agricultural land more intensively (Kirk 1999).

Fire regimes will be modified as climates change (UNEP 2002b). Climate anomalies (drought and above-average rainfall) affect vegetation dynamics and, therefore, the timing, size and intensity of fires. Southern Africa is subject to large interannual variability in rainfall associated with the El Niño/Southern Oscillation (ENSO) phenomenon, sea surface temperature cycles, and land-atmosphere feedbacks (Nicholson 2000). Climate predictions indicate that Africa could be 2-6°C warmer in 100 years though how rainfall will change is more uncertain (Hulme et al., 2001). At present, climate predictions are limited by the absence in climate models of, among others, any representation of regional changes in land cover and atmospheric loadings of dust and aerosols from biomass burning, all of which are related to fire regime (Hulme et al., 2001). Although decadal and multi-decadal forecasting of the African climate remains uncertain, seasonal forecasting has advanced rapidly both in its development and application. Forecasts of upcoming seasonal rainfall patterns are regularly issued and used, particularly on the probability of drought, widely associated with a warm phase of the ENSO, and above-average rainfall, associated with cold ENSO events. Seasonal forecasts may be useful in anticipating the threat of extensive fires. Studies are needed to predict changes in the pattern of burning under changing climate conditions and should be linked to studies of the use of fire for improved land management.

There are no adequate data on the occurrence, size distributions, or trends in fire numbers or areas burned annually in southern Africa that meet the information needs of policy and decision makers (Frost 1999). Local fire information exists for some national parks, forests and other conservation areas, but these lands are not representative of the region as a whole because they are subject to specific fire management policies and are largely protected from the influence of ordinary people. Remotely sensed fire data, in conjunction with ancillary information, are needed to understand the basic relationships between fire, population and land use, and to develop a better understanding of the underlying processes (Frost 1999). Such information would allow policy makers, planners and managers the opportunity to understand fires in their environmental, economic and social contexts, and to formulate their responses accordingly (Stocks et al., 2001).
The utility of satellite fire products to meet these information needs has not been demonstrated widely. Active fire detection products from a number of satellite sensing systems (Section 4) do not provide reliable information on the spatial extent and timing of burning, as clouds may preclude hotspot detection and because the satellite may not overpass when burning occurs (Eva and Lambin, 1998; Roy et al., 2002a). Algorithms that use multi-temporal satellite data to map areas affected by the passage of fire – burned areas – are less constrained by these problems and are now receiving more attention. The limits of detection (e.g., the completeness of combustion, the spatial distribution and size of burns, and the ability to map fire-affected areas beneath unburned overstorey vegetation) have yet to be established, however. The timeliness, accuracy, utility and interpretability of satellite fire products must be considered before resources are invested in their uptake. This is particularly true in southern Africa where immediate concerns for social and economic development, education, agricultural production and food security limit the resources available for managing the environment. The potential for international and regional co-operation in sharing information and resources, and undertaking joint actions to understand and control fires to minimize their adverse effects, needs to be explored.

5.3 FIRE AND LAND MANAGEMENT IN THE BOREAL REGION

Fires represent an important disturbance in the boreal forest regions of Canada, Alaska, and Russia, but little of the boreal forest in Fennoscandia burns (Figure 4; McGuire et al., 2002). In Canada and Alaska fire occurs throughout the boreal forest, with about 94% of the area burned in Canada located in the boreal forest. Over the last 50 years, an average of approximately 2.3 million ha burns annually in the North American boreal forest, with the amount of annual area burned ranging between approximately 0.5 and 8 million ha (Amiro et al., 2001; Kasischke et al., 2003).

![Figure 4. Patterns of historical annual area burned with vegetation distribution along the International Geosphere Biosphere Programme’s Terrestrial Transects in Alaska, Canada (Boreal Forest Transects Case Study – BFTCS), Scandinavia (Finland), and Russia (East](image-url)
In northern Eurasia, most of the fires occur east of the Ural Mountains. It is estimated that large fires account for 90% of the area burned in central Siberia (Ivanova et al., 2002). In comparison to Alaska and Canada, official fire statistics available from Russia suggest that area burned in Russia is relatively much less with only approximately 1 million ha burning annually and that maximum annual area burned is less than 3 million ha (Kasischke et al., 2003). However, analyses based on satellite data estimate that 11.7 million ha burned in 1987 and that 13.3 million ha burned in 1998 (Conard et al., 2002; Kasischke et al., 2003). More careful analyses of fire frequency in Russia suggests that fire frequency in Siberia is higher than in Alaska and Canada (Figure 4; Shvidenko and Nilsson, 2000; Shvidenko and Goldhammer, 2001; McGuire et al., 2002), although fires in Siberia tend to be low-intensity ground fires as opposed to crown fires in Alaska. Thus, for boreal forests outside of Europe and European Russia, official statistics of fire in Russia probably represent a significant underestimate of burned area and available data suggest that between 0.5 and 1% of boreal forest burns annually with high rates in Siberia, for example, along the Yenisey River (Figure 4; McGuire et al., 2002).

During the 1970s and 1980s, the fire frequency in some parts of northwest North America increased substantially, and since the 1980s the amount of area burned annually is approximately double the area that burned prior to 1970 (Kurz and Apps, 1999; Stocks et al., 2000; Podur et al., 2002; Kasischke et al., 2003). While analyses of the official fire statistics in Russia suggest that fire has almost doubled between the 1970s and 1990s (Kasischke et al., 2003), this trend clearly needs to be verified as it may be also affected by changes in reporting practices. Because the boreal forest contains approximately 30% of the global terrestrial carbon inventory (McGuire et al., 1997), an increase in fire frequency throughout the boreal forest and peat lands has the potential to release large amounts of carbon into the atmosphere.

The degree to which increased fire frequency has the potential to release carbon in the boreal forest depends, in part, on fire severity (Conard and Ivanova, 1997; Michalek et al., 2000; Isaev et al., 2002). Fire severity is a function both of the vegetation type and fuel structure and of the effects of weather on fire behavior. Carbon release may vary four to ten-fold between low severity surface fires and high severity crown fires (Conard and Ivanova, 1997; Shvidenko and Nilsson, 2000; Conard et al. 2002; Kasischke and Bruhwiler, 2002). Many forests in Russia to the east of the Ural Mountains and into central Siberia are dominated by Scots pine \((Pinus sylvestris)\) with an open understory. These forests are characterized by surface fires in which trees usually survive because of thick bark, although high intensity surface fires or crown fires can occur under severe conditions. In severe fire years, crown fires may make up to 50% of the total area burned in Scots pine forests of Russia (Belov, 1976). The extensive larch forests of central Siberia and the Far East also have a fire regime often dominated by surface fire. In contrast, the dense forests of spruce, fir, and pine species such as Jack pine or Siberian pine, which occur on colder and wetter sites in Siberia, the Russian Far East, and North America tend to have fire regimes dominated by intense, stand-replacing crown fires, which result in near total mortality of the canopy trees.

Forest fire management in both Canada and Alaska relies on the Canadian Forest Fire Weather Index (FWI) System (Canadian Forestry Service, 1987) to understand the current fire danger and state of fire alert required to ensure adequate
resources are available for daily fire suppression activities. In Canada, daily fire monitoring since the 1970s has relied on aircraft for detecting fires, a monitoring technique that has replaced traditional fixed fire towers in all provinces except Alberta. In Alaska, aircraft have also served an important role in both fire detection and suppression. Alaska was one of the original test regions for aerial suppression techniques in the United States, and an aerial detection program was formally organized by the Bureau of Land Management (BLM) in 1973. The first lightning detection system was established in the Fairbanks district in 1975. Additional detection units and upgrades have provided statewide detection capabilities. Aerial detection now focuses daily reconnaissance flights in areas where ground lightning strikes were detected.

In Russia, forest fire management relies on the Nesterov Index (Nesterov, 1949) and the Moisture Index (Vonsky et al., 1975) to understand the current fire danger and state of fire alert. These indices are not as elaborate as systems currently being used by Canada and Alaska, which makes them less reliable in predicting fuel moisture and potential fire behavior, which is important information for planning fire suppression efforts. Because of the enormous area of the Russian boreal forest, the Russians are currently using operational systems that rely on AVHRR data for real-time detection of active fires and for near-real-time mapping of burned areas. In Canada and Alaska remote sensing is primarily used to analyze post-fire burn characteristics, but operational fire management does not currently rely on remote sensing for the monitoring of active fires. The use of remote sensing for active fire management in Canada and Alaska will likely not be attractive until the technology proves itself to be superior to current monitoring systems with respect to its role in suppression efforts.

Humans affect fire regimes in the boreal forest through the ignition of fires, the suppression of fires, and land use that alters the spatial distribution of fuels. In North America, human-caused ignitions are more prevalent in the more heavily populated southern area of the boreal ecozone in Canada and along the road network in Alaska. It is estimated that humans are responsible for 70% of ignitions in western Russia and 67-93% of ignitions in eastern Siberia (Ivanov 2002). Although humans are an important source of fire ignition in boreal forests, lightning is responsible for the majority of area burned. In North America, only 10 - 20% of the area burned in boreal forest is due to human ignitions. Poor burning conditions at the time of ignition and fire suppression efforts are responsible for the low amount of burned area from human-caused fires. Also, the human alteration of fuels across the landscape may play a role in promoting longer fire cycles near populated areas and along transportation networks (see Chapter 9).

While fire suppression efforts in Canada are the direct responsibility of individual provinces, territories, and national parks, the Canadian Interagency Forest Fire Centre has been established to coordinate suppression efforts among agencies. Close to 50% of the area burned in Canada results from fires that receive no suppression because of their remote location, low amount of property value at risk, and efforts to accommodate the natural role of fire in boreal ecosystems. In Alaska, the Alaska Fire Service (AFS) is an interagency coordination network for detecting, monitoring, and suppressing fires. In general, the Alaska Department of Natural Resources is responsible for fire suppression efforts on state lands and on lands in close proximity to the road network, while the BLM AFS is responsible for fire suppression efforts on most federal lands and areas located away from the road system. In contrast to the lower 48 states, in Alaska there is a high degree of coordination and cooperation among state and federal fire management and suppression forces. Economic problems hamper fire management in Russia and
many fires receive no suppression because of remoteness and lack of adequate resources for suppression. In Russia fire suppression efforts are organized by region through the Russian Aerial Forest Protection Service. Additional suppression forces can come from local forestry districts.

Climate and fire interact on intra-annual, interannual, and longer time scales. The fire season in the boreal forest runs from late April through October, with most of the area burned occurring in June and July. The pattern of early summer burning occurs soon after snowmelt during a dry time of the year, which sets the stage for low fuel moisture in early summer. The weather generally becomes wetter as the summer progresses, so that fires generally are extinguished by the end of summer. In contrast to temperate and tropical areas, where the fuel of concern is aboveground biomass, soil organic matter in the forest floor of boreal forests is important for the lateral spread of fire. The importance of soil moisture is highlighted in the Canadian FWI system, which relies on tracking moisture in the surface fine fuels, the top duff layer and the compact organic material at the bottom of the duff layer.

While fire activity has been documented to be highly variable from year to year in specific regions of the boreal forest (Kurz and Apps, 1999; Stocks et al., 2000, 2003; Amiro et al., 2001; Conard et al., 2002; Kasischke et al., 2003), it has only recently been elucidated that fire in the boreal forest region as a whole has enough interannual variability that it may play a role in the interannual variability in the growth rate of atmospheric CO$_2$ (Langenfelds et al., 2002). Analyses have established that interannual variability in area burned in the boreal forest of North America is linked to interannual variability in climate with area burned generally greater for higher growing season temperatures and lower growing season precipitation with long sequences of days that are characterized by low rainfall and relative humidity below 60% (see Chapter 9). Large fire years in central Siberia appear to be associated with drought, and the fire cycle is more frequent in regions that experience more drought years per decade (Ivanova et al., 2002). The Canadian Forest FWI has proven useful in analyzing variability in fire frequency among regions of the boreal forest (Flannigan et al., 2001), and analyses of FWI for climate change projections suggest that climate change has the potential to substantially increase fire frequency throughout much of the boreal forest (Wotton and Flannigan, 1993; Stocks et al., 1998; Flannigan and Wotton, 2001; Flannigan et al., 2001).

6 Conclusions

These three case studies from contrasting regions show the common need for improved monitoring of fires, leading to accurate annual inventories of fire distribution and timing and area burned. Accurate inventories of burned area when combined with improved spatially explicit models of fuel load will provide important inputs to improve current emissions estimates and understand the annual variability of fire and long term trends and their impact on the atmosphere. Developing validated global annual fire and burned area inventories of known accuracy is a major objective for the LCLUC program.

A strong case needs to be made for elevating fire to a higher priority in the design of operational satellite monitoring systems. To secure the necessary long-term global fire inventories needed to identify trends in different fire regimes beyond the large signal of interannual variability, the new technologies developed and tested by NASA will need to be adopted and supported by the operational
agencies. Similarly, changes in infrastructure or procedures of these agencies will be needed to integrate new information into the current resource management systems. The NASA LCLUC program can then redirect its resources to new research and development to fill gaps in our current understanding. For example, satellite-borne lasers may provide improvements in our measurement of fuel loads, data assimilation combining weather data and satellite vegetation condition may provide improvements in our mapping of fire risk. New sensors may provide a better characterization of fire intensity.

In all three regions fires need to be viewed in their environmental, economic and social context. Integrated regional assessments of fire need to be developed based on an understanding of both the causes and the effects of fire. Improved mesoscale climate modeling is needed to provide more accurate predictions of future fire conditions. Similarly, realistic projections of population dynamics and land use are needed to indicate areas of changing fire potential and hazard. Realistic projections of changes to fire regimes remains a research challenge for the LCLUC program, which will require integrated modeling combining physical and social processes. Providing reliable fire information in a timely fashion and in a form interpretable by users will increase the use of satellite data in support of policy making. This can be achieved by providing an assessment of current fire patterns and their underlying causes to evaluate the effectiveness of existing fire policies and/or by helping to craft new policies based on actual fire occurrence and recent fire history. Providing the research underpinning to improve fire and resource management and the associated land use policies is a long-term goal for the fire research in the NASA LCLUC Program.

7 References


LAND USE AND FIRES


CHAPTER 20

LAND COVER / USE AND POPULATION

RONALD R. RINDFUSS1, B. L. TURNER II2, BARBARA ENTWISLE3, STEPHEN J. WALSH3

1University of North Carolina at Chapel Hill, Department of Sociology, Chapel Hill, NC 27599
2Clark University, Graduate School of Geography and George Perkins Marsh Institute, Worcester, MA 01610
3University of North Carolina at Chapel Hill, Chapel Hill, NC 27599

1 Introduction

Land use change is important from a variety of perspectives. It is of interest in its own right. Processes like urbanization, the expansion of a frontier, land degradation, or recreational uses of land have long been of theoretical and research interest. Population growth, and the pressure it puts on land use and agricultural practices, has been an issue since Malthus’ classic essay and has been central to the thinking of such 20th century scholars as Hawley (1950), Davis (1963) and Boserup (1965, 1981). More recently, however, it has been global change issues that have driven interest in land use change (Lambin et al., 1999; de Sherbinin 2002). As the relationships between land use change and climate change are better understood, it has become self-evident to broader scientific constituencies that population processes influence land use change, and that land use change can, in turn, influence population processes.

The three processes that determine population growth are fertility, mortality and migration, with the well-known relationship for population change over time:

\[ \text{Pop}_t = \text{Pop}_{t-1} + (\text{births} - \text{deaths}) + (\text{in-migrants} - \text{out-migrants}) \]

At the global scale, the migration term drops out; but below the global scale migration is likely to be the most dynamic. Given that mortality is moderately low in most countries and that there is a universal value to decrease mortality as much as possible, mortality is unlikely to have a strong effect on land cover and land use change (LCLUC) in the foreseeable future and we will not discuss its likely effects on LCLUC here.¹

Another important set of population considerations involves its composition and distribution. From the perspective of LCLUC, perhaps the most important aspects of population composition and distribution are changes in urbanization and in household size. Since urbanization is discussed elsewhere in this volume we will not cover it here except to note that the indirect or consumptive demands on the land by an increasingly urbanized population has local to global consequences. We do address the issue of household size

---

¹ It is important to note however, that the initial stages of mortality decline are characterized by substantial reductions in infant and young child mortality. When this occurs, from the perspective of population growth, it is the demographic equivalent of an increase in mortality because those infants and children who survive under the new mortality regime go on to eventually have children of their own.
and the extent to which it has an effect over and above the effect of population size. Put differently, for a given population size, does it matter if people are aggregated into a relatively small or large number of households.

Our focus is on the present, and we assume stability of governments, institutional arrangements, and the like. We assume that no natural or human induced calamity will have a major and non-linear impact on population size or processes. Yet we note that the two sites studied by this chapter’s authors have experienced sharp population declines in the past, leading to substantial land use change. Nang Rong, in Northeast Thailand, was under the influence of Ankor until the empire collapsed sometime in the fifteenth century (Rooney 1999; Rindfuss et al., 2003). While little is known about that period, it appears that the land in Nang Rong, and indeed the entire Ankor empire, reverted to a condition wherein there was limited human influence. The southern Yucatan peninsular region was home to the ancient Maya civilization. At its apex, about AD 900, the region maintained population densities in excess of 90-100 people/km2. This population collapsed to virtually nothing, and the region remained largely unoccupied until the late nineteenth century. Substantial occupation and use in modern times awaited development programs in the middle of the twentieth century (Turner, Geoghegan and Foster 2003).

Of course, as has long been recognized (Hawley 1950; Ehrlich and Holdren, 1971; various authors in Jolly and Torrey, 1993), the effect of population change likely interacts with such other factors as affluence, life style and technology. A recent meta-analysis of case studies (Geist and Lambin, 2001, 2002) re-emphasizes this point. They reported that multiple causes or “cause clusters” are needed to explain most cases of deforestation in the tropics. Even though demographic variables were frequently found to be important, typically it was only in conjunction with other factors. Angelsen and Kaimowitz (1999) similarly conclude that no simple set of explanatory factors work across all scales. As such, numerous interactive factors influence land use patterns that involve complex linkages that are locale and scale dependent. Waggoner and Ausubel (2001) also recognize this complexity when they examined the extent to which population growth will lead to forests being converted to crop land. In addition to population, other factors such as income, diet patterns, non-food crops (e.g. cotton and tobacco) and yield need to be considered. This chapter focuses on population and LCLUC issues, but it is important to remember that population factors are unlikely to be acting in isolation from other forces, and that feedback mechanisms and critical thresholds may produce non-linear relationships.

Even though we primarily focus on population changes affecting LCLUC, we begin by discussing how land use might affect population change to emphasize that the population-LCLUC relationship goes in both directions. We then discuss the relevance of scale, noting that the population-LCLUC relationship can change from one scale to the next. Then we discuss fertility and migration effects in turn, followed by the special case of population effects on protected areas. Finally, we discuss whether it is the number of people or the number of households that influences LCLUC, recognizing that they are highly correlated with one another.

---

2 Rudel (2001) shows that the expansion of forest land between 1935 and 1975 across Southeast U.S. was related to increased agricultural yields elsewhere.
2 Changing Land Use Affecting Population Change

The point here is straightforward, but we provide a section for it to underline its importance. There has been a tendency in the literature and in informal discussions to assume that population growth is the problem and that it adversely impacts LCLUC. This chapter emphasizes how population change might influence land use, but it is also important to remember that changes in land cover and use can impact population processes. Perhaps the earliest arguments along these lines came from Malthus who posited that populations grew as arable land was converted to agriculture. North America provided considerable data for his arguments. As the frontier pushed westward, agricultural land, and hence food, became abundant. He argued that this, in turn, led to high rates of natural increase, that is, the birth minus death balance was positive and large.

Other examples exist. Land that has become degraded will trigger a variety of population responses. Consider a natural example. Lightning induced fire can negatively affect the landscape. After the fire has died down, those who lived in or near the forest are likely to out migrate. If they stay, the degraded landscape might affect their ability to secure needed foodstuffs, which, in turn, might lead to higher morbidity and mortality.

There is also increasing evidence that land cover change can affect climate, which in turn could affect agricultural productivity, which could affect population processes. For example, the extensive deforestation in Southeast Asia appears to have affected the timing and quantity of the monsoon (Kanae, Oki, and Musiake 2001). Without irrigation, the lower rainfall and greater variability makes wet rice cultivation subject to greater uncertainty. In turn, this could lead some to migrate.

Land use also affects fertility. For example, not only may high fertility encourage deforestation, fertility may be high on the frontier where forest land is abundant (e.g., VanLandingham and Hirschman, 2001). As another example, not only may fertility affect the size of landholdings, landholdings may affect fertility (e.g., Cain 1985; Stokes et al., 1986; Maglad 1994). Although this chapter focuses on the effects of population on land use, it is important not to lose sight of these feedback effects.

3 Scale and the Population-LCLUC Relationship

A fundamental aspect of the relationship between aspects of population and LCLUC is scale. As already noted, if the scale is global, migration falls out of the basic population growth equation. But the scale issue is more fundamental than this simple algebraic example; as has long been recognized in geography, scale is likely to affect the nature of relationships between population and LCLUC. Hierarchy theory, which was developed in general systems theory and incorporated into ecology to describe the structure of ecological systems through their spatial and temporal organization (Forman and Godron, 1986), can be used to frame this discussion. The theory advocates the interrelationship of scale, pattern, and process, and states that process operating at the scale above a given or “characteristic” scale serve as context, whereas the processes operating below the “characteristic” scale describe the mechanisms (O’Neill et al., 1988). Hierarchies also can change with time thereby emphasizing issues of resilience and adaptability of systems.

Fundamental to studies of land cover and land use (LCLUC) dynamics and human behavior are the interactions between landscape composition, spatial organization,
and time (Dale et al., 1993). In studies where LCLUC characterization is achieved through remote sensing technologies by mapping state (e.g., cover type classes) and/or condition (e.g., plant biomass levels) variables, the spatial scale of observation is critical to the defined pattern (Allen and Walsh, 1996). Depending upon the instantaneous field of view (IFOV) of the sensor system, the spot size of the sensor array may extend from fine to coarse grains and the extents may range from local to regional. For instance, Landsat Thematic Mapper (TM) has a spatial resolution of 30 x 30 m for its optical channels, whereas the NOAA (National Oceanic and Atmospheric Administration) AVHRR (Advanced Very High Resolution Radiometer) sensor has a spatial resolution of 1.1 x 1.1 km. This difference in the grain size of the satellite remote sensing system used to spatially partition the landscape into picture elements or pixels generates very different spatial views of landscape composition and patterns of LCLUC variation (Frohn et al., 1996). The areal extent of the imagery is also important for characterizing patterns. If the “footprint” is too small patterns may be incoherent and unorganized because of the limited “view” of the landscape. On the other hand, “footprints” too large may amalgamate a number of fine-grained patterns that could obscure general relationships.

Often in LCLUC studies, mapping the landscape for a single “snapshot” in time is inadequate and what is needed is the development of an image time-series to characterize LCLUC dynamics and to create change images that capture important elements of landscape flux (Lambin and Strahler, 1994; Lambin 1996). In such situations, choosing a system with the necessary temporal depth is important, and so too is the temporal resolution of the sensor system for characterizing LCLUC patterns on intra- and inter-annual periods and for decadal views as well (Crews-Meyer 2001). Aligning the image time-series, for instance, to longitudinal surveys and important social (e.g., an economic crisis) and/or environmental (e.g., a drought or flood) events adds relevance and richness to the imagery as a consequence of the population-environment links across space and time (Moran et al., 1994).

Linking people and the environment through space has been challenging because of differences in the spatial grains and extents, the mobility of people, their discrete representation across the landscape, and the fact that people are generally not captured directly through remote sensing systems, but indirectly through their “imprints” on the landscape. On the other hand, the environment is generally stationary, represented continuously across the landscape, and the environment can be assessed in a number of fundamental ways through remote sensing systems. While remote sensing systems partition the landscape into pixels and record spectral responses for them that are generally affected by in-situ characteristics such as soils, drainage, settlement patterns, and disturbance regimes, people can range over wide geographic areas affecting local and regional land uses and extending their influence on the land through time by land tenure systems, kinship ties, social networks, and land management schemes that further link people and place. Even when resources or energy move from one to cell to another thereby affecting in-situ conditions, routing approaches allow the characterization of “flows,” for example, flows of water from nearby, upstream cells and flows of soil as a consequence of erosion or other forms of land degradation. Flows of people that affect local land uses at specified locations can also be accommodated, but it is not straight-forward and it is subject to significant data needs and spatial-temporal linking issues.

Scale dependent studies have indicated that there is no “correct” scale to study the landscape and to link population to space-time observations of LCLUC change. But there
are ranges of spatial and temporal scales that offer insight to patterns and processes believed to be important in understanding how, why, and when the landscape has changed, and how landscape form may be related to landscape function. Such studies have sought to examine how people, place, and the environment are interrelated across a range of spatial and/or temporal scales (Walsh et al., 1997, 1999). In general, they found that population variables are more strongly associated with LCLUC at finer scales, and less likely to have a statistical association at coarser scales.

4 Fertility and Land Use Change

Relationships between land use change and fertility flow in both directions. High fertility and population growth may contribute to such outcomes as deforestation, agricultural extensification, intensification, land fragmentation, soil degradation, and urbanization. Pressures on the land caused by high fertility and growing household size may lead families to change their land use strategies, to change the timing and number of births, or to move out of the area (Davis 1963). Thus, the effect of fertility on land use change will depend on the appeal of the other responses and can only be understood in a larger context. The effect will also depend on social, economic, and cultural context. Factors such as land availability, tenure arrangements, markets for crops, availability of non-agricultural employment opportunities, technology, and the like will have conditioning effects. There is no single common effect of fertility on land use; nor is one expected. As a review of the literature shows, the effect of fertility on land use varies from place to place and time to time.

The hypotheses underlying expected effects of fertility on land use change in rural areas is that larger families require more food to subsist when children are young, have more labor with which to cultivate land when children are older but still living in the home household, and require more land when children grow up and need a place of their own to farm. Effects in the short term may differ from effects in the longer term. There is an important life cycle aspect to how households use land (Entwisle et al., 2001; McCracken et al., 2002), and the timing and quantum of fertility is an important aspect of the household life cycle.

When land is abundant, households may adjust to increasing family size and subsistence needs by enlarging the area under cultivation. Often, this involves the conversion of forests to agricultural uses (Paulson 1994; Foster et al., 2000; Umezaki et al., 2000). However, to say that fertility and rural population growth may fuel deforestation in some settings is not to say that it is a major driver of deforestation in all settings (e.g., Moran 1994). Even within a given setting, the importance of fertility may vary depending on characteristics such as accessibility and population density (Rosero-Bixby and Palloni, 1999).

There may be both short and long run effects of increasing family size on the expansion of land under cultivation. In the short run, land may be cleared to provide for increased subsistence needs. In the long run, additional land may be cleared as larger cohorts of children grow up and establish households of their own, a phenomenon fairly common in frontier environments. The longer-term effect of high fertility would operate through increased rates of household formation, a topic addressed later in this chapter. Although fertility has declined in most parts of the world, a legacy of high levels in the past
is a continuing growth in the numbers of young people coming of age, forming their own households, and using land for dwelling units and for some type of productive activity (e.g. farming, some sort of manufacturing activity or office work). To see this consider Figure 1, which shows population pyramids for India 2000 and 2025, with an assumption that fertility declines to just above replacement level.

Figure 1. Population pyramid for India 2000 and 2025. Expectation of life at birth in 2025 is 70.5 and the total fertility rate is 2.2.

By comparing the two figures it can be seen that the shape is becoming rectangularized at the younger ages. This is a classic example of the implications of declining fertility on the age distribution. Thus, even though reduced fertility leads to diminished growth of the base (ages 0-4), the legacy of past fertility leads to substantial increases in the numbers of men and women entering their 20s and 30s many years after the decline in fertility.

Another possible response to increasing family size is intensification, which may occur in a variety of ways. Societies dependent on swidden agriculture may increase frequency of cultivation. Studies of shifting cultivation do find that population growth
related to increases in fertility shorten periods of fallow (Saikia 1998; Umezaki et al., 2000). Increasing frequency of cultivation was of particular interest to Boserup, who first proposed the intensification hypothesis, but intensification may occur in other ways as well, including multiple uses of the land, the use of fertilizers, pesticides, herbicides, and high-productivity seeds, and improved irrigation (Mortimore 1993). It may also involve the use of more marginal lands for purposes other than those for which they are ideally suited. For example, in Nang Rong Thailand, lands marginal for paddy rice cultivation were brought into production as the region was settled and populated (Welsh 2001; also see Cruz 1996). Intensification is more likely to occur when the availability of new lands is limited. A related longer-term outcome is fragmentation. Over time, existing plots of land are likely to be divided into smaller and smaller pieces (e.g., Lavely and Wong, 1984), and unless new crops or techniques are introduced that support larger families on smaller and smaller plots, the fragmentation may ultimately drive rural families into poverty. This process, of course, depends on such institutional factors as land tenure systems and inheritance practices, as well as land resource endowments.

Fertility has consequences for the household’s use of land that it owns or claims. Fertility may also have consequences for lands held in common. Growing local populations may put pressure on lands formerly held in common. Common lands may be “claimed,” enclosed, and used for new purposes. Common lands may also be used for traditional purposes, but more intensively. The Chitwan Valley in Nepal provides an example of the effects of household size on environmental consumption, that is the use of natural resources, either directly or indirectly. Axinn et al. (2002) find a positive effect of household size on the collection of wood from local forests for cooking. Follow up work by Pearce, Axinn, and Chaudhary (2003) indicates that household size must be considered not only in terms of numbers of children, but also in terms of household structure (e.g., extended or not).

There are more complicated effects as well. Fertility has declined in many parts of the world over the past several decades, but remains higher in rural than urban areas of most places. Yet, over the next few decades, growth in urban places is expected to outstrip rural areas, and rural-urban migration is one of the reasons why (United Nations 2002). Rural-urban migration will transfer part of the impact of rural fertility to urban places and play a role in the conversion of land to urban uses. We now turn to migration.

5 Migration and Land Use Change

Many would argue that, among the demographic processes, migration is the most likely to affect land cover and use. The reason is fairly straightforward. Large numbers of people can move to new locations in a relatively short period of time. They might migrate for economic reasons or for purposes of being closer to natural and other amenities. Alternatively, they might be refugees migrating because of political, ethnic or religious persecution. Whatever the reason, the speed with which migrants can move makes it a potentially volatile factor affecting land use in both the sending and the receiving areas.

3 By their very nature, refugees and forced migration streams have been difficult to study. Even estimating their numbers is problematic. A recent workshop at the National Academy of Science estimated that there may be as many as 50 million in the world (Reed, Haaga, and Keely, 1998). Unfortunately, little is known about their impact on land use or land cover.
The effect of migration on land use has a long history within the population field. Malthus is usually remembered for his arguments about how unchecked fertility will lead to increased poverty and scarcity of foodstuffs, and ultimately increased mortality. But, as Bilsborrow (2002) has recently noted, Malthus also argued that rural farm populations would respond to population growth by out-migrating in search of land, and hence expanding the frontier.

Population is not randomly scattered across the globe. People who migrate are attracted to places for a variety of reasons, including good farm land, climate, and attractive natural settings (e.g. Bilsborrow 2002). The same is true for urban areas which commonly are located at transportation break points and places of opportunity. Small and Cohen (1999), using the Gridded Population of the World data set, find a strong relationship between population densities and elevation. Population densities are highest at low coastal elevations and then in topographic basins adjacent to mountain ranges. Population densities decline rapidly with increasing elevation and distance from coastlines.

Expansion into frontier areas is perhaps the best studied impact of migration on LCLUC, with the continental U.S. being one of many examples. In recent years most of the attention has focused on the tropics, particularly expansion into tropical forests. Numerous studies point to the role of migration (e.g. Hecht and Cockburn, 1990; Bilsborrow and DeLargy 1991; Smink and Wood, 1993; Sader et al., 1997; Moran and Brondizio, 1998; Wood and Skole, 1998; Walker et al., 2000). These migration stories are not simple (e.g. Perz 2002), and it is important to recognize both facilitating and push factors are involved. In many places the facilitating factors have included roads built for other purposes (such as timbering or oil exploration) and governmental policies (either to encourage migration to frontier regions as had been the case in Brazil or to discourage migration into protected areas as is exemplified in numerous countries). While we cannot run the experiment to see what might have happened if roads had not been built, many would argue that migration would have been substantially reduced in the absence of facilitating road building. Similarly, it is important to consider the push factors at place of origin including population pressure as a legacy of prior fertility levels.

Deforestation associated with in-migration has undoubtedly received the most attention, but regrowth can occur after the initial migration and reforestation has been associated with out-migration. For example, there is evidence from the Brazilian Amazon that there is a migrant family cycle relationship to deforestation and then secondary regrowth (Brondizio et al., 2002; McCracken et al., 2002). There have also been arguments about “forest transitions,” that is after a period of rapid deforestation and population increase, followed by a period of stability, a turnaround in forest cover occurs. Using countries as his units of analysis, Rudel (1998) shows that this reforestation turnaround is associated with urbanization and industrialization. The processes facilitating reforestation at the sub-country level likely include rural to urban migrations; the southeast US during the middle of the 20th century provides an example (Beale 1964; Rudel and Fu, 1996).

Migration associated with reforestation illustrates the obvious: migration involves an origin and a destination. Most of the work linking migration and land use involves effects on destinations. Origins are also affected, and we need to better understand how

---

Another component of migration in many countries, particularly developing countries is circular migration, and, as far as we know, there have not been studies examining the relationships between circular migration and land use. Indeed there is not complete agreement on the definition of circular migration, sometimes referred to as seasonal migration. For our purposes we mean movements that are relatively short duration, repetitive (hence the term “circular”), and planned by the individual to be brief. Migrant farm workers are one example. Perhaps the most common example in developing countries are rural residents who go to urban areas in search of jobs during the dry (that is, the “off”) agricultural season. Circular migration can be internal or international (e.g. Hugo 2003). Quite frequently such seasonal migration is not captured in standard migration data sources (Hugo 1982; Cordell et al., 1996), and so estimates of its volume are lacking for most places. But there is enough scattered evidence to suggest it is quite common. From the perspective of land use, circular migrants are likely affecting more than one locality, in many cases one rural and one urban. They are likely keeping rural areas under cultivation and perhaps contributing to sprawl in urban areas.

6 Population on the Fringe of Protected Areas

The “development-environment” frontier has long captured the attention of various scholars in regard to diffusion, development, and population-resource dynamics. Global environmental change and sustainability studies raise a new frontier, that related to protected areas, as the lands within their borders harbor the last vestiges of “the wild” and provide a multitude of ecosystem services, from carbon sinks to biotic diversity to surface water regeneration (Raven 1997; Daily et al., 2000). Advances of the population-development frontier on protected areas worldwide are identified as leading to “hot spots” of loss of biotic diversity (Achard et al., 1998; Bruner et al., 2001; Homewood et al., 2001; Liu et al., 2001), degradation of ecosystem function, and declines in the well-being of parks and reserves (Achard et al., 1998; Bruner et al., 2001; Hanson and Rotella, 2002) and trumpet a call to arms to find proper design and management systems for viable protected areas, including international biosphere status, mosaics of protected areas arranged in corridors to promote the movement of biota, and co-management programs predicated on sustainability criteria (e.g., West and Brechin, 1991; Ghimire 1994; Neuman 1997; Berkes and Folke, 1998; Brandon and Redford, 1998; Zimmerer and Young, 1998; Oates 1999; Vanclay et al., 2001; Verísimmo and Chochrane, 2001). These concerns follow from the recognition that the lands, more often than not, have been long occupied and used, even if ephemerally, and in many cases are witnessing population and development pressures within those protected areas that contain people and along their borders or in adjacent “buffer zones” otherwise. While much attention has been given to protected areas under threat within the developing world, the build up of population and rural-to-recreational-urban changes in land-uses around protected areas possess threats in the developed world as well (e.g., Achard et al., 1998; Bruner et al., 2001; Hanson and Rotella, 2002). This build up on the fringes of protected lands is related to the desire of people to want to live or vacation on the edge of biologically diverse and/or scenic lands, and hence is related to the migration discussion above (Hansen et al., 2002).

At least two major human-environment issues are related to this new population-
frontier issue: (i) pressures degrading landscapes within and surrounding the protect area, affecting the functioning of ecosystems within the protected areas and (ii) co-management practices in which the basic functioning of the protected area requires human presence and use. Regardless of co-management strategies, development pressures adjacent to or within protected areas affect the flora, fauna, and ecosystems being protected. In contrast, some protected areas require human presence or use to maintain the ecosystem under protection (e.g., burning of savannas in eastern Africa).

A simple view may divide the central forces at play between population and development pressures in the non-western and western worlds respectively. The reality, of course, is far more complex. For example, the lands surrounding Yellowstone National Park in the U. S., have succumbed to significant occupation as well as recreational uses, fragmenting the landscape with housing, roads, and other infrastructure (Hansen and Rotella, 2002). In contrast, the lands adjacent to the Serengeti-Mara reserves of Tanzania-Kenya have not only witnessed an increase in permanently settled people but encroachment of large-scale commercial farms in key breeding habitats (Homewood et al., 2001). The critical point is that population and development pressures tend to coincide and analysis must account for the synergies between them.

These pressures alter land-covers, some of which can be tracked through the use of remotely sensed imagery and analysis. Major land-cover changes within a protected area have obvious implications for the biota present, especially those involving forest preserves. The size and pattern of forest remnants is an important element to be considered (Malanson 2003). For example, agricultural expansion driven by population growth within the Wolong Giant Panda Reserve in China has reduced forest cover and hence the requisite bamboo needed to preserve the panda (Liu et al., 2000). Increasingly, however, research illustrates the role of land changes outside the reserve, often in the so-called “buffer” zone on the biota within the reserves. For example, population driven land-changes around the Serengeti-Mara reserves in eastern Africa affect the long-standing movement of large wildlife in and out of reserves, and commercial wheat farming has taken away significant portions of the breeding grounds of the wildebeast (Homewood et al., 2001), potentially affecting their numbers and the ecological consequences of reduced wildebeast. In another example, population-settlement growth along the high biotic diversity riverine habitats has disrupted the winter habitats supporting the Yellowstone bird population, and the fencing of lands in general around the park plays havoc with the movement of bison and elk (Hanson and Rotella, 2002).

Some of these examples, of course, raise flags about co-management schemes, intended to create a means of protecting nature and people, such as the Mesoamerican Biological Corridor (Miller and Chang, 2001). Some (Bruner et al., 2001; Vanclay et al., 2001) have concluded that such systems are not working well in regard to tropical forest preserves, in part because such systems are highly sensitive to most human uses and increasing population drives disturbance. Alternatively, in co-evolved landscapes and ecosystems—those in which a long-standing human presence has helped to generate the land-covers in question—the loss of human activity may change land-covers and reduce wildlife, as may be occurring in parts of the Serengeti.
7 Changing Households and Land Use Change

So far we have discussed the two likely dynamic components of population growth: fertility and migration. Another aspect of population is how its members (individuals) are organized into households. Even though most authors examining the impact of population on land use or other aspects of the environment concentrate on population size per se (e.g. de Sherbinin 2002), there are reasons why it might be more important to look at how people are organized into households rather than simply examining the number of people. Before discussing the impact of people vs households, we paint a broad overview of trends in household size.

While there are exceptions, of course, the rising affluence associated with the transitions from agricultural to industrial to post-industrial societies has been accompanied by a shift to smaller household sizes, that is, the number of individuals living in a household (MacKellar et al., 1995; and also compare the various U.N. Demographic yearbooks). There are a variety of reasons. From a demographic perspective, other things being equal, declines in fertility will lead to smaller household sizes. Many forms of rural to urban migration result in one generation residing in an urban area and another generation living in the rural origin area. Many countries have experienced increases in divorce (e.g. Bumpass 1990), and this often turns one household into two. And, in some countries a stage in the life course has emerged when children leave the parental household but have not yet formed their family of procreation (e.g. Goldscheider, Thornton, and Young-Demarca, 1993) and this frequently results in the creation of an additional household. There are also broader reasons. The multi-generation households traditionally preferred in many societies are often the source of interpersonal friction – sometimes vividly captured in fictional accounts of traditional family life. When rising affluence permits mobility from multi-generational households, splitting into smaller units is typical. Lesthaeghe (1983; Lesthaeghe and Moors, 1995) has argued that there has been a rise in the value of individualism, particularly in Western Europe, but perhaps elsewhere. As individualism becomes more highly valued, so might privacy and hence declining household size.

Declining household size, in turn, can affect land use through a variety of mechanisms. Other things being equal, declining household size means demand for more housing units. Typically these units will be spread horizontally across the landscape rather than vertically, possibly leading to urban sprawl and/or more dwelling units occupying the rural landscape. More dwelling units leads to more demand for building materials, affecting forest cover and perhaps open-pit mining. Smaller households also translate into less efficient use of various resources, and this could also impact land use.

There has been relatively little research that has linked changing household size to changing land use. The data demands to do so are formidable, particularly given the interconnections and feedbacks between changing household size and changing land use. One study in Northeast Thailand (Entwisle et al., 1999) showed that the effects of population change when expressed as change in household size had a larger impact on the conversion of land for use in upland crops (cassava, corn, sugar cane and the like) than when expressed as counts of individuals. Corollary evidence (Liu et al., 2003) suggests that decreases in household size have a negative impact on biodiversity, which, in turn, is strongly related to land use.5

---

5 MacKellar and his colleagues (1995) have illustrative projections suggesting that changes in household size have a bigger impact on CO2 emissions than simple changes in population counts.
8 Conclusion

We finish by briefly emphasizing several points. First, it is important to re-emphasize the complexity of the population-LCLUC relationship. Causality can and does go in both directions, with feedbacks. Scale, both geographic and temporal, matters. Relationships evident at one scale might not be clear at another. It is not just population, but how the population is distributed over the landscape and organized into households.

Second, when considering population effects on land use it is critical to remember that it is population in context. This has long been recognized, but frequently overlooked. The effect of population on LCLUC depends on numerous factors, including technology, the social organization of the population, the material status of the population, and its consumption patterns.

Finally, while the field has produced impressive advances in our understanding of population-LCLUC relationship in recent years, it is also humbling to realize how much we still do not understand. Sometimes it is because the questions have not yet been asked. Sometimes it is because appropriate data and/or methods are not yet available. Whatever the reasons, there is still plenty of room to improve our understanding.

9 References


Section IV   Methodological Issues, Modeling
CHAPTER 21

TRENDS IN LAND COVER MAPPING AND MONITORING

CURTIS E. WOODCOCK, MUTLU OZDOGAN

Department of Geography, Boston University, 675 Commonwealth Avenue, Boston, MA 02215 USA

1 Introduction

It is an interesting time for mapping and monitoring of land cover using remote sensing. There are exciting new kinds of maps derived from remote sensing at a variety of spatial scales. For example, there are a number of new kinds of maps of land cover becoming available globally. Following in the footsteps of the global land cover products derived from the 1992 1km AVHRR time series (the IGBP Discover Map (Loveland et al., 2000) and the University of Maryland Land Cover Map (Hansen et al., 2000)), two new products have been developed. One is the GLC2000 Land Cover Map (Bartalev et al., 2003) made using data from the SPOT4-VEGETATION sensor. Another is the MODIS land cover map, made using a time series of data from the MODIS sensor (Friedl et al., 2002). All of these maps are categorical in nature and include classes roughly at the level of biomes. There are differences between the legends of these maps, but their basic nature is similar. A somewhat different set of products are based on the idea of continuous fields, with a good example being percent forest cover (Hansen et al., 2002). These maps are also global and provide a different perspective on land cover.

At the same time that these global maps were being produced using coarse resolution imagery (meaning imagery of approximately 1km or coarser), new maps at high spatial resolution over large areas were made. The best example in this regard is the use of Landsat imagery to produce a 30m land cover map of the United States, referred to as the MRLC land cover map (Vogelman et al., 2001). For the first time it is now feasible to acquire and process the large amounts of high resolution data necessary to map large areas in great spatial detail.

2 Land Cover Mapping

Given the exciting nature of recent developments in land cover mapping, questions arise concerning related trends and issues for the future. We offer the following postulates to reflect our views of key trends:

2.1 POSTULATE 1

The lessons learned in the processing of coarse resolution imagery should improve the mapping of land cover at fine resolutions, and vice versa.
To date, the development of land cover mapping using fine resolution imagery has proceeded somewhat in parallel with the use of coarse resolution imagery. This has occurred primarily due to the extreme differences in the nature of the imagery available. At coarse resolutions, AVHRR imagery had few spectral bands and high temporal resolution, so temporal trajectories of NDVI became the primary information source available for mapping land cover. This focus on the temporal domain was effective given the interest in mapping broad classes reflecting biomes that are primarily controlled by climate. Similarly, the nature of the imagery available at fine resolutions, such as from the Landsat satellites and the SPOT HRV sensor, dictated a focus on the spectral domain for land cover mapping. Most applications relied on a single date of imagery and used the differences in the spectral signatures to differentiate land cover types. Particularly prior to the launch of Landsat 7, single images were the norm because integration of additional images was expensive and the rate of acquisition of images was such that it was often difficult to find multiple images of the same area in the same year (Goward et al., 2001).

The nature of the imagery available for land cover mapping changed dramatically with the launch of the Landsat 7, TERRA and AQUA satellites. At coarse resolutions, the MODIS sensor on TERRA and AQUA has greatly improved spectral resolution relative to AVHRR imagery, as well as finer spatial resolutions (250m and 500m) (Justice et al., 1998). Thus, the opportunity now exists to explore the use of spectral and temporal signatures for mapping land cover using MODIS imagery and thus there is much to be learned about the use of data from the spectral domain from the community with experience analyzing Landsat data. Another area where much could be learned from the fine resolution community concerns accuracy assessment. It is well established for maps of local areas derived from fine resolution imagery that they are incomplete without statistically valid accuracy assessment (Congalton and Green, 1999; Stehman, 2000). Efforts at global mapping would benefit from more attention to accuracy assessment. To date, the only statistically valid accuracy assessment for a global land cover map was for the IGBP-DIS map (Scepan, 1999). It is easy to recognize the problems of trying to randomly sample the entire world from a logistical perspective, yet improved accuracy assessment of global land cover maps would significantly enhance their value and use. In essence, they are not truly complete without an accuracy assessment.

At fine spatial resolutions, the recent innovation is not the nature of the sensor but the frequency with which images are being collected. The ETM+ sensor is similar to prior Landsat TM sensors in terms of numbers and location of spectral bands. However, the acquisition plan used for Landsat 7 and the updating of the processing stream makes images available more frequently and more quickly after collection than was previously possible. Thus, it is more feasible to use multidate imagery at high spatial resolutions. There have been a number of examples showing the value of at least multisseason imagery (Wolter et al., 1995; Mickelson et al., 1998; Oetter et al., 2001), but there is much more to be learned with respect to temporal data. The experience and lessons learned with coarse resolution should improve this effort with fine resolution data.

Another way the fine resolution community could learn from the experience of the coarse resolution community concerns compositing of imagery. Coarse resolution imagery is routinely composited to remove the effect of clouds. But the history of the use of fine resolution imagery is to try to wait for imagery that is free of clouds. For
large area mapping, particularly in many tropical areas with chronic high cloud cover, the frequent acquisition of Landsat imagery provides the opportunity to begin to composite Landsat data. Such an approach might greatly improve the ability to map large areas in the tropics using fine spatial resolution imagery.

2.2 POSTULATE 2

_Creative use of remote sensing inputs as well as ancillary data sources will improve the mapping of land cover more than further development of classifiers and mapping algorithms._

The primary components of the land cover mapping process are _inputs_ and _methods_. There have been substantial improvements in land cover mapping methods over the past decade, particularly the development and adoption of methods based on decision trees (Friedl and Brodley, 1997; Hansen et al., 2000), artificial neural networks (Benediktsson et al., 1990; Carpenter et al., 1997) and spectral mixture analysis (Smith et al., 1994; DeFries et al., 2000). The value of the development of these methods should not be minimized. However, it is our contention that there is significant opportunity to improve mapping of land cover through creative use of the wide variety of imagery now available.

2.2.1 A Wealth of Imagery

There is an unprecedented variety of imagery now available with the potential to make contributions to land cover mapping. We have already mentioned the improvements in Landsat imagery associated with the launch of Landsat 7. Similarly, we have already mentioned the MODIS sensor, which provides coarse resolution imagery at high temporal resolution with a valuable selection of spectral bands. Another interesting new sensor, MISR (Diner et al., 2002), provides optical imagery collected at a variety of view angles providing the opportunity to explore the use of directional data in land cover mapping. There is considerable theory indicating the value of directional data for recovering information on vegetation structure (Li and Strahler, 1992), but there has been relatively little use of directional data for land cover mapping (Hyman and Barnsley, 1997). Another sensor on TERRA is ASTER (Yamaguchi et al., 1999), which provides both multispectral optical and multispectral thermal imagery at high spatial resolutions. The value of multispectral thermal imagery for land cover mapping has yet to be explored. Finally, there are two new commercial sensors providing very high resolution (on the order of meters) imagery of the Earth’s surface, IKONOS (http://www.spaceimaging.com) and QUICKBIRD (http://www.digitalglobe.com). The net effect is that multispectral imagery at a variety of spatial resolutions, view directions and temporal repeat times is now available. Many improvements in land cover mapping will come from improved use of this imagery.

2.2.2 The Potential of Image Fusion

Associated with the wide variety of images now available is the possibility of combining images from a number of sensors, or image fusion. Such an approach can take advantage of the strengths of different kinds of imagery, with the result being an improvement over what is possible with a single sensor. As a community, we are still
in a fledgling state of development with respect to image fusion and great advances remain to be achieved.

Two brief examples help illustrate ways image fusion can help land cover mapping. Liu (2002) used classifications of land cover derived from Landsat imagery to train an artificial neural network to map the fractions of various land covers within MODIS pixels. The strength of this approach is that using a single Landsat scene, it is possible to train an algorithm that can be applied over a much larger area using MODIS imagery. Since the land covers of interest (forest, agriculture, water, barren) are easily discerned in Landsat imagery, it affords a straightforward approach to provide training data for a complex mapping process with MODIS data. A second example illustrates the value of MODIS data to help map land cover at local scales with Landsat imagery. For a project for the NASA LCLUC Program, Ozdogan et al., (2003) found a strong disjunction in the seasonal patterns of growth for rain-fed and irrigated agriculture in Southern Turkey. The clear picture of these seasonal patterns provided helpful guidance regarding the optimum timing for fine resolution Landsat imagery, which was necessary to monitor changes in irrigation over time. Many other examples of the benefits of image fusion are emerging, and this exciting approach to remote sensing will grow and yield substantial benefits over time.

3 Land Cover Monitoring

Monitoring of environmental change is rapidly becoming one of the most common and valuable uses of remote sensing. The archive of imagery that now exists, primarily from Landsat and AVHRR, allows for monitoring a wide variety of changes in the environment at a number of spatial and temporal scales. This exciting capability for tracking the changes in landscapes resulting from both human and natural causes is providing a better understanding than we have ever had of how our planet is changing.

Associated with the increasing use of remote sensing to monitor land cover and land use change are questions about how to best utilize the information content of the imagery. One unmistakable trend is the tremendous proliferation of methods being employed for detecting change using remote sensing. Examples abound in the literature (Lambin and Strahler, 1994; Bruzzone and Prieto, 2000; Yamamoto et al., 2001). One cause of this proliferation of methods is a substantial degree of innovation in methods that is undoubtedly improving the results of analyses. A second possible cause of this proliferation of methods is that it is often unclear what the best available methods are for any given application. The literature has many examples of the use of remote sensing for monitoring a wide variety of environmental changes. However, in most cases these examples are guilty of the one place, one time syndrome. In this syndrome, investigators report on the results of their attempts to use one method for one problem in one place at one time. This approach is entirely understandable because the investigator is often studying one place at one time and is not necessarily interested in the methodological dimensions of the problem. One unfortunate result of the one place, one time syndrome is that it is difficult to generalize regarding which methods are most useful for which applications. In this context, some generalizations are possible:
There is not a single method of change detection in remote sensing that is best for all kinds of imagery, environmental change processes and locations. Given this second condition, is it possible to begin to sort applications of change detection such that it may be possible to begin to categorize change monitoring problems? Or, stated another way, there is a need for a way to think about the relationship between the methods available and different kinds of environmental change such that it is easier for people to figure out which methods make the most sense for their applications.

3.1 TOWARD A TAXONOMY OF CHANGE PROCESSES FROM A REMOTE SENSING PERSPECTIVE

A taxonomy of change processes requires consideration of a variety of factors, but when complete allows for generalization regarding how to link sensors, analysis methods and various kinds of environmental change processes.

3.1.1 The Nature of Environmental Change: Categorical vs Continuous Change.

One reasonably direct way of sorting environmental change processes concerns the measurement scales used to measure the change. Some changes are inherently categorical (or measured using a nominal scale), particularly when monitoring land use and land cover change. One example is urbanization, where land that was previously used in other ways (possibly agriculture or natural vegetation) is converted to urban use. Another example is deforestation, where areas that we previously forest are no longer.

The other alternative is for changes in amounts or concentrations of some property of landscapes which can be measured as a continuous variable. One example is the change in LAI or cover of a vegetation canopy over time. Another example is the magnitude of forest mortality following a drought (Collins and Woodcock, 1996) or the severity of a forest fire (Rogan and Yool, 2001).

While the distinction between categorical and continuously measured change is simple in the abstract, it is worth noting that in practice the differences are often blurred. For example, the magnitude of change in a surface property (such as burn severity) is often reduced to ordinal categorizes (high, medium and low).

3.1.2 The Nature of the Desired Information: does space matter?

Some applications of environmental monitoring are focused on knowing where the change is occurring, while others are more directly interested in knowing how much change is occurring. In essence, is the desired result a map, or area estimates? In some instances both are desired, and it is often necessary to make a map in order to estimate the area of change. But this fundamental distinction is often not recognized and can have implications regarding the selection of methods.
3.1.3 The Nature of the Intended User

Trying to match applications, sensing needs and methods also depends on the nature of the intended user of the results. One way of categorizing the users of the results of monitoring of environmental change using remote sensing follows:

- L (local resource management/planning)
- I (invading scientists)
- N (national scale planning/management)
- G (monitoring global change, science and subsequent policy)
- W (early warning: droughts, el-nino, …)

The context of local resource managers is straightforward and differentiated from national planners where the focus is more on policy than actual management of individual parcels of land. The category of *invading scientists* implies scientists from some other part of the world studying environmental change not because they are responsible for mitigating the problem, but because they are interested in better understanding the problem. Monitoring of environmental change in support of global change science implies a degree of generalization and scope generally not of interest to those involved in local land management or often even national planners. Early warning applications are identified here separately due to the importance of the timing of information, as further discussed below.

It is interesting to note that for the same kind of environmental change the nature of the intended user can have significant implications. The example of monitoring deforestation helps illustrate this point. Local resource managers may be primarily interested in maps of deforestation, while global change scientists may be more concerned with area estimates of the amount of deforestation and not so directly concerned with its precise location.

3.1.4 Time Scales of Environmental Change

There is a wide range of time scales over which environmental change processes occur. However, monitoring via remote sensing is limited essentially to the following:

- E (events, i.e. days to months)
- A+ (annual, one to several years)
- D (decades)

Examples of *events* could range from monitoring the extent of flooding to mapping forest fires to the effects of a volcanic eruption. Remote sensing has been used more commonly at the time scales of years to decades. Typically deforestation and urbanization have been monitored over these longer time periods. It is worth noting that some kinds of environmental change cannot be monitored over time scales as short as a few years. Take the case, for example, of climate change. Time periods more on the order of decades are required to differentiate climate change from interannual variability.

The existing archive of remote sensing imagery limits the time periods over which many changes can be monitored, but that time period continues to increase. The value and existence of this archive provides a very strong case for continuity in satellite measurement programs. The availability of a strong historical archive of imagery is at least as valuable as the ability to make observations today. The argument can also be made that it is priceless, as it could never be recreated.
3.1.5 Frequency of Observations Required
A second temporal dimension to the sensing requirements of environmental change concerns the frequency of observations required. Again, the existing suite of satellite sensors limits the opportunities significantly, particularly when this consideration is combined with spatial resolution requirements, as discussed below. For monitoring change in terrestrial environments, the following frequencies need to be considered:

- D (daily – or hourly to weekly)
- S (seasonal)
- E (endpoints)

By far the most common approach for monitoring environmental change using remote sensing is to use two dates of imagery, one to mark the start of the monitoring time period and one to mark the end. This endpoints approach is so common that it is frequently presumed to be the case for most environmental monitoring efforts. However, other approaches can be required in certain situations and for some applications. An example of an application requiring daily to weekly frequencies might be the need to monitor change in vegetation amount (LAI, for example) in response to climate change. To be able to detect changes in a variable as dynamic as LAI, it may be necessary to have daily to weekly measurements of LAI over many years.

3.1.6 Spatial Resolution Requirements
Different kinds of land cover change have different requirements for the spatial resolution of imagery. The categories below reflect the primary sensing capabilities currently available:

- H (high resolution <10m)
- M (medium resolution 10-100m)
- C (coarse resolution >500m)

Note the gap in resolutions between medium and coarse resolution. This gap is interesting, as much imagery at 250 m from MODIS is becoming available and future sensors (such as MERIS) will make extensive use of this resolution. What will be interesting to see is whether analysis of change at these resolutions will prove useful for use at local scales and extend the kinds of work done locally with Landsat imagery in time and space, or whether it will simply improve regional to global scale analyses by providing improvements over current capabilities with such sensors as AVHRR and SPOT4-VEGETATION.

3.1.7 Geographic Coverage Requirements
It has been common in remote sensing to link the concepts of spatial resolution and coverage. This is natural to some extent as the geographic coverage of individual images has been related to the spatial resolution of sensors. However, it has been demonstrated that it is possible to analyze many images in automated ways for land cover mapping and monitoring. Hence, it is important to separate the concepts of spatial resolution and geographic coverage, as some environmental change processes may require detailed spatial resolution, yet large area coverage. The following list differentiates common geographic extents for monitoring environmental change using remote sensing:

- S (sites, up to 1,000 km^2)
- L (local, i.e. areas up to 100,000 km^2)
- R (regional, roughly between 100,000 and 1,000,000km^2)
3.2 A CONCEPTUAL MODEL FOR THE CHANGE DETECTION PROCESS

Figure 1 provides a conceptual model for the various kinds of steps used in the change detection process. It is limited to the endpoints paradigm, which we have noted as being dominant – particularly with respect to land cover change.

The intent of this conceptual model is to try to illustrate which kinds of methods can be substituted for each other, and which fill different roles in the processing stream. One important note here is that not all of these steps are necessary in all situations. The best case in point concerns radiometric preprocessing, which in this case primarily concerns atmospheric correction of images such that the images from different dates are directly comparable. A number of authors have indicated the necessity of atmospheric correction as a component of change detection, and in many cases this is true. However, depending on the steps that follow, atmospheric correction may not be necessary. For example, in multidate classification of two dates of a single Landsat scene, it may not be necessary to atmospherically correct the images prior to classification (Song et al., 2001).

When attempting to determine which methods to use for change detection, it is often best to begin at the right side of Figure 1 and work backwards. This approach requires thought about the nature of the expected results, as discussed above. Is the desired result area estimates or a map, and if it is a map, is it measuring categorical or continuous change? Based on this decision, it is then possible to determine which methods are best for relating image change to environmental change. For example, some of the methods indicated are best for producing categorical maps of change, such as density slicing and classification. Others are better suited to continuous measures of change, such as regression and spectral mixture analysis. Still others can be formulated in either way, as is the case with decision trees and artificial neural networks.
Depending upon the method selected to relate image change to environmental change it may be advisable to transform the original multidate imagery. For example, to produce a categorical map showing areas of deforestation using density slicing, it might be helpful to do something like image differencing of NDVI, or use PCA to search for a component that indicates change in vegetation amount. There are many methods of transforming multidate images in an attempt to isolate change. Change vector analysis is a popular and effective method that is commonly used. More analytically complex methods (such as PCA or Multitemporal Kauth-Thomas) may be necessary if the desired environmental change is a subtle effect in the images. Simpler methods may be sufficient for more obvious kinds of changes. Note that some methods of relating image change to environmental change rarely require transformation as an initial step. This is true of spectral mixture analysis and classification.

Based on this conceptual model, it is possible to begin to determine which methods make the most sense to use together, and which combinations do not. For example, for image classification it is rarely necessary to correct for atmospheric effects. Similarly, for many kinds of image classification (such as maximum likelihood) it is unnecessary to transform the multidate image prior to classification. However, most uses of spectral mixture analysis require atmospheric correction of imagery such that the data are converted to the units of reflectance.

There is a strong relationship between the nature of the end user and the nature of the desired information on land cover change. As the users move along a gradient of extremes from local managers to global change scientists, the nature of the desired information tends to change from maps to area estimates. Thus, to study the same kind of environmental change in the same area, different kinds of end users might be best served by different processing methods!

4 Concluding Thoughts

It is an exciting time for both mapping and monitoring land cover. The wide variety of imagery available is enhancing our ability to map land cover and monitor changes in the environment. Much benefit will be gained from increased use of imagery from multiple sensors, or image fusion. Also, improved use of the spectral domain for coarse resolution sensors and the temporal domain for fine resolution sensors should improve land cover mapping.

The taxonomy of change processes and the conceptual model for monitoring change using remote sensing provide a framework for helping people determine which methods might be most appropriate for their application. In particular, it should help people recognize the similarities and the differences between examples in the literature and their applications.
5 References


CHAPTER 22

LINKING PIXELS AND PEOPLE

RONALD R. RINDFUSS1, STEPHEN J. WALSH2, B. L. TURNER II3, EMILIO F. MORAN4, BARBARA ENTWISLE1

1University of North Carolina, Department of Sociology, Chapel Hill, NC 27599
2University of North Carolina, Department of Geography, Chapel Hill, NC 27599
3Clark University, Graduate School of Geography and George Perkins Marsh Institute, Worcester, MA 01610
4Indiana University, Center for the Study of Institutions, Population and Environmental Change, Bloomington, IN 47405

1 Introduction

Integrated land-change science (Turner 2002) seeks to join remotely sensed (pixels), biophysical (terrestrial), and social science (people) data. The history of joining these three fundamentally different data types is remarkably short. The effort focused on the social science-remote sensing data beyond photogrammetrics has only emerged over the last decade or so. The organizers of the National Academy of Science volume People and Pixels (Liverman et al., 1998), which was designed to provide illustrations of studies that joined the two types of data, had to scramble to find a sufficient number of experts and research to fill a workshop and the resulting volume. Indeed the careful reader of that volume might wonder about the extent to which some chapters actually link social science and remotely sensed data. The paucity of robust pixel-people studies is unfortunate given the increasing need for such linkages as global environment change, biocomplexity, and sustainability science turn to questions of the coupled human-environment system.

This chapter reviews some of the issues that arise when joining remotely sensed and social science data. The focus is methodological, not substantive. The goal is to identify, describe, and review methodological challenges, recognizing that the solutions will be driven to a large extent by a researcher’s substantive questions and scientific goals. As noted, the history of joining data on pixels and people is short. Hence it is highly likely that some key questions have not even surfaced, a point to which we return in the conclusion of the chapter. Also, we do not address potential ethical issues that might arise when joining remotely sensed and social science data except to note here that ethical issues definitely do arise and researchers need to be careful about them (see discussion by Rindfuss and Stern 1998). Readers must remember that the assessment offered here is a start and not a finish.

The chapter opens with perhaps the most fundamental question that researchers interested in joining people and pixels must face: where to begin? There is no necessary

1 We speak in terms of pixels because the typical study uses spatial data from satellites, which comes in raster form. We recognize, of course, that one might want to link to spatial data, such as soil maps, that might come in the form of polygons. The general points we are making about linking to raster data would also apply to polygons.

parallel between social units and land units. Moreover, coverage of one does not necessarily guarantee coverage of the other. The first section discusses the implications of the starting point, land or people. Then, linking relations (e.g., ownership, use, or access) are addressed. The need for temporal depth presents challenges on both sides of the land-people equation, and these are discussed next. We then turn to challenges associated with the joining of diverse disciplines, which is often a necessary part of joining people and pixels. The chapter concludes with an overview of new topics and challenges that will need to be addressed as the field develops. To date, researchers linking people and pixels at finer scales have tended to focus on land use in rural areas, especially in frontier environments; there is a need to encompass urban areas as well.

2 Starting Point: People or Land?

In linking pixels and people, an early decision that researchers must face is whether to start with people and try to link with pixels, or start with pixels and try to link to the people affecting those pixels. Where one starts is determined by the research question, the data available, and in many instances the disciplinary orientation of the researcher. Where one starts is consequential for the kinds of statements that can be made, and the kinds of conclusions drawn.

There are important differences between the types of data typically available for land and those typically available for people. Remotely sensed data provide continuous coverage of land within some predetermined boundaries, from relatively small areas such as a county or a watershed to relatively large areas such as a continent or the entire globe. In contrast, social science data tend to refer to discrete units: individuals, households, organizations, and nations. Social science data are rarely global in reach. When they are, such as world demographic and economic data, they are aggregated (e.g., by the United Nations) from country supplied data for countries that vary considerably in their quality. The scales at which social science data are reported varies, but often refer to administratively defined units such as counties, provinces, or countries. Social science analysis typically draws on data referring to a smaller unit of social organization such as a household. More often than not, these scales are inconsistent with those of land studies. Although governments make and enforce policy for administrative territories such as nations, provinces, and municipalities, there is no spatial unit that corresponds to an individual, household, or business. At finer scales, the challenge is to marry continuous land data with discrete data on social units (Rindfuss et al., 2003a).

Considerable experience exists in linking biophysical and spatial data sets to one another (e.g., overlays of different coverages in a GIS) and in linking social survey and administrative data sets (e.g., merging data sets for different units of observation for hierarchical statistical analysis). For each, there are critical theoretical and technical issues, such as accurately overlaying the pixels from the various images with each another to make sure they cover the same land units, deciding the successor to a household, business, religious group or governmental body when the original unit subdivides or fissures, or linking a person to one context (e.g., residence) when they actually live their lives in multiple contexts as determined by residence, work, seasonal work, vacations, and the like.

While acknowledging that these and other complex issues exist in creating linked data sets within domains, they will not be discussed here. Rather, we focus on the issues arising out
of moving across these types of data sets to link people and pixels.

Suppose we choose land as our starting point. Starting with land offers advantages in research where there is a strong biophysical component being considered as part of the research design, and where the quality of the people data is of poor quality. We might choose a biophysical unit, such as a landscape or watershed, or a territorially defined political unit such as a county or district thereby permitting a link to detailed biophysical data and/or detailed census data collected for that land unit. If we choose a biophysical unit, the challenge is to identify and assemble data for the relevant social groups. There is no social unit that parallels a landscape or watershed, for example. Starting with land provides the potential for continuous coverage within the boundaries of the land units chosen, but the people and organizations to which links are made may not represent a coherent collection of people or organizations.

We might also sample land units, and hence their associated pixels in a satellite image. This is not uncommon at relatively fine scales, such as a field plot. Examples include Moran et al. (2003) and Walsh et al. (2003). Once a sample has been drawn, the research design challenge becomes (a) the identification of the most proximate decision makers for the land sampled (e.g., through ownership records at a local government office), (b) locating and obtaining relevant information (e.g., from those owners directly, or indirectly through records), and (c) locating and obtaining relevant information about land cover, land use, soils, and other biophysical factors that will permit interpretation of the remotely sensed data. The link to people might be through some type of individual or group interview, or it might involve administrative records, such as land transfer data. A potential problem is that the ability to acquire the relevant information may depend on the characteristics of the people involved. For example, owners of the sampled land units are likely to vary with respect to a variety of characteristics, such as type (individual, household, extended kin group, business, limited partnership, NGO, or government), location (on the land unit, near the land unit, quite distant from the land unit including in another country), and economic or wealth status. These and other characteristics are likely to be related to the ability to locate and obtain the necessary data. Others things equal, we expect businesses, and wealthier and more distant individuals to be the most difficult entities from which to obtain data. To the extent that non-response is significant, and to the extent that it is selective in the way we have just described, researchers need to consider the implications for bias. Put differently, if non-response is substantial and selective, then the resulting data set is likely to be biased.

Starting by sampling people (individuals, households, organizations, and so forth) and then linking them to the land parcels that they own or use also raises problems related to coverage and data quality. The link to the land might be made through respondent interviews, administrative data, or informants. Again, there will be issues of respondent cooperation and quality. The availability of data on land may depend on the characteristics of that land. For example, often there is more ambiguity about ownership and use rights to

---

2 Technically, pixels refer to the smallest unit of observation in a remotely sensed image. For linguistic ease here, we also use “pixels” to refer to land units as viewed through a rasterized scope.

3 In using administrative records, the issues of respondent non-response (e.g., Groves 1989; Groves and Cooper, 1998; Freedman, Thornton and Camburn, 1980; Lessler and Kalsbeek, 1992) recede, but one then needs to worry about the coverage and quality of the administrative data. People supplying the administrative data might have an incentive to not be completely forthcoming (e.g., to avoid taxes). Administrative records are also frequently out of date.
forest lands than to lands in other coverage. An additional important point is that when people are sampled, the land parcels linked to the sampled people are unlikely to be contiguous; instead they are likely to be a patchwork of parcels surrounded by other unlinked parcels. The surrounding unlinked parcels might belong to people who were not interviewed or might result from missing data in administrative records. Alternatively, the surrounding unlinked parcels might belong to people (individuals, households, or organizations) that were not in the sampling frame used by the study. For example, the sampling frame might be all individuals, households and organizations in a given district. Thus land owned or used by those outside the district would not be linked or included in the study. No matter what the reason, starting with people is likely to lead to a patchwork of linked land units which is unlikely to have any ecological or environmental coherence, and may introduce bias from the perspective of the land that is not linked.

As more case studies accumulate, the implications of starting with land or people will need to be carefully examined. Some studies in tropical forests begin with land (e.g. Moran et al., 2003; Walsh et al., 2003), while others begin with nucleated villages or households (Rindfuss et al., 2003b; Turner and Geoghegan, 2003). This decision is often a result of the way the settlement pattern itself has come into being, and/or the availability of relatively good quality property boundary data that allows for a one-to-one link between a social unit (household) and a land unit (parcel). In communities, such as ejidos, where land is held in common this is a less likely option (unless customary behavior has resulted in customary recurrent use of the same land area by individual families). If one draws a sample of land units, say field plots, except under unusual circumstances, the owners of those plots will not yield a representative sample of households living in proximity to those plots. For example, landless households, as well as households renting land from absentee landlords, would be excluded. Similarly, a sample of social units will not generally yield a representative sample of land units. For example, not all land is owned or used by households. Descriptive statements about land based on a sample of social units will not generally agree with descriptive statements based on a sample of land units. Similarly, descriptive statements about social units based on a sample of land units will not generally agree with descriptive statements based on a sample of social units. Clearly, making explicit how a researcher is making the link, and with which unit they are beginning is an important starting point for synthesis efforts in order to determine whether cases are indeed comparable. When attempting to draw lessons and generalities from the entire set of case studies (see Geist and Lambin 2002), it will be important to critically address the extent to which differences in the base sampling strategy affect the outcomes that are included in the synthesis of case studies.

3 Selection of the Linking Relation

Linking pixels and people requires a linking relation. Suppose one wishes to join a village and a village territory. The linking relation might be an administrative or tax boundary, and the linking unit, the administrative village. Suppose one wishes to link a household and a field plot. The linking relation might be ownership, and the linking unit the owned land parcel. Linking relations can be complex. Consider, for example, the people who have the most proximate decision making power over a given collection of pixels or a land unit. In an area where land is clearly titled, the “person” (e.g., individual, household,
business, NGO or government) with the most proximate decision making power is the land owner, even though other individuals or institutions may exercise decision-making power, such as those who might rent the land, zoning that regulates permissible uses of the land, or the influence of market forces (i.e. demand) for products from the land. When land is not clearly titled, identifying those with the most proximate decision making power can be problematic.

Just as there are numerous units of observation on either side of the people-pixel divide, there are also numerous potential linking relations such as managing the land, owning the land, renting the land, using the land for recreational purposes, and so forth. In addition, linking at one level does not guarantee links at other levels. Within each domain, there may be a hierarchy. Individuals are embedded within households, within villages, within administrative units all the way up to the country or region itself. Field plots are embedded within patches, within landscapes, within regions, and so forth. Much effort has been put into linking households and field plots, as described below. However, aggregating household-plot links will not generally yield a meaningful link between, say, villages and landscapes. In other words, linking people and pixels at one point in the hierarchy does not create parallelisms elsewhere. This section of the chapter briefly reviews some of the issues that arise with respect to linking relations, beginning with the “easiest” case of direct ties between land managers and specific land parcels.

In cases of land ownership and customary usufruct rights, individual households or land managers can be linked specifically to land parcels; this is also true of ranching or stocking systems linked to private ownership (e.g., Archer forthcoming). Farming systems with direct ties between land managers and specific land parcels would seem to alleviate the linkage problem. Even in this situation, the user-parcel relationship can be quite complex, however. In some cases, multiple, discontinuous parcels are cropped by one household and a single parcel may be cropped cooperatively by several households. Rindfuss et al. (2003b) describe a multi-pronged approach to the matching of households and plots in this situation involving household interviews, group interviews, parcel boundaries superimposed onto aerial photographs. In other cases, spatially continuous parcels may be used in a crop-fallow cycle in which forest succession makes parcel demarcation difficult (Moran et al., 2002). Turner and colleagues (2001) treated this problem in southern Yucatán by sketch mapping entire parcels in various stages of succession and linking them to satellite imagery by obtaining a GPS reading of the parcel. Even more complexity is introduced in those cases where parcels in different stages of succession are rented or borrowed by neighbors from the land “owner.” In these cases, the household from which social information is drawn may not be that which determined the cropping strategy. Laney (2002) addressed this problem in Madagascar by undertaking household surveys in which the history of land borrowing was detailed, and then linked to the overall imagery of the lands enjoined by the village. Further, in some settings, land ownership might be something that the owners do not want divulged in an interview setting (e.g. Rindfuss et al., 2003b).

Another variation is to move away from the most proximate land users and obtain data from a more highly aggregated unit, such as a village or a district. One version of this strategy uses village-level focus groups or rapid rural appraisals to gather data at the village

---

4Clearly, a complex, multi-purpose research design will want to have links to these more distal decision makers. Many of the issues we address for the most proximate decision makers will also apply to the more distal ones.
level (e.g. Fox et al., 2003) linked to satellite imagery of the village lands. Alternatively, census level socioeconomic data collected for political or administrative units, usually at a higher level of social aggregation such as a county or municipality, are linked to remotely sensed data of the same unit (Rosero-Bixby and Palloni, 1998; Wood and Skole, 1998; Geoghegan et al., 2001; Seto and Kaufmann, 2003). Both of these strategies have the virtue that they reduce the costs to the researcher of collecting the social science data. Focus group information, however, typically produces “reduced” form data, often qualitative only, while census data reduces the questions of the study to those that are possible based on preexisting data collected for other purposes. The drawback to both approaches, but especially the census/district level design, is the now classic argument that there is no necessary relationship between relationships at the district level and relationships at the individual, household or organization level (Robinson 1950). Relationships between land and people may be scale-dependent (Walsh et al., 1999). Further, the census approach has a scale problem noted by McCracken and his colleagues (2002). The size of areal units is inversely related to population density, in order to protect the confidentiality of individuals and households. In many rural areas, where density is low, the size of the areal unit is typically larger than the analyst might want, because county or district units are spatially very large in low population regions.

As just explained, even in farming systems with direct ties between land managers and specific land parcels, linkage can be a problem. In many parts of the world land is not clearly titled or is held as common property or with common access rights. At the extreme, herders move their stock from place to place on a daily or seasonal pattern, seeking adequate feed and water, and creating special problems for linking people and pixels. BurnSilver and her colleagues (2002) addressed this issue in the Kajiado District, Kenya, by joining herders on their daily grazing paths, using a GPS unit to record the spatial location of these paths, providing a direct link of the people to the specific lands used. In contrast, Robbins (1998) made this linkage by combining village-level data collection with the common access areas used by the villagers. In short, when land is not clearly titled, identifying those with the most proximate decision making power can be even more problematic than when land is clearly titled. As a result, the investigator designing a study for such a site will need to know the formal and informal rules governing land use in the area, through preliminary field trips and/or careful reading of the relevant research literature.

A different type of problem emerges because not all people who have an effect on the land live on or close to the land. Companies that may exploit the land (e.g., logging, mining) may be based at some distance. Likewise for policy-making bodies. But even at a finer scale, individuals who impact the land may not live on or close to it. The general problem is how to identify these distal decision makers, obtain data from them, and then make the link to specific land parcels. To illustrate the issue we consider tourists, but note that the problem is far more general than just tourists.

Many of the places where people go for vacations tend to be fragile environments, such as coasts, areas rich in flora and/or fauna diversity, and areas on the edge of places that are geologically spectacular (such as Yellowstone or the Grand Canyon in the U.S.). Indeed eco-tourism is being suggested as a way to save ecologically important regions, particularly those in developing countries. While the link to hotel owners and eco-tourism operators might seem straightforward, the hotel owners are probably located quite some distance from the site, thus having very little direct feedback on the impact of their
decisions on the landscape. National and international governing bodies will frequently have land use rules and regulations applied to such areas, but may sometimes have limited ability to enforce the rules meant to protect such habitats, or lack the involvement of local people to ensure enforcement. Commonly, people will live in these reserves or protected areas—sometimes legally and sometimes in violation of existing regulations. Liu and his colleagues’ work illustrates the importance of understanding human influences in such settings—in his case the Wolong Nature Reserve (Liu et al., 2001). Liu’s work also illustrates the difficulty in establishing micro links in reserves and protected areas (Liu et al., 2003). Even more difficult is linking remotely sensed data, and changing land cover, for these fragile and vulnerable areas with vacationers who come for a week or two. The vacationers who use the fragile landscape for short periods are affecting land cover change, but they are likely to live and work quite some distance from the land being affected. No less complex are the links at larger scales, such as the broad impacts of eco-tourism in parks across the world. In such cases, a complex array of variables may be relevant to understanding how short-term visitors, coming in large numbers sequentially over a short vacation period, have aggregate impacts much greater than comparable numbers living in a similar area on a year-round basis. While this is a research question that has not been addressed to date, it is a reasonable hypothesis on the basis of the greater likelihood of having feedbacks to the latter, and an absence of those same feedbacks to the short term visitors.

4 Temporal Depth and Associated Complexities

People-pixel links, of course, are far more complex than land users-parcel relationships. They also involve temporal dynamics and other change complexities, both in land-cover and causes of use-cover change. Consider, for example, the geophysical complex light phenomenon known as reflectance. A parcel is made up of biophysical factors such as soils, water, plants, and buildings. Each of these reflects light in its own particular way, and these data fluctuate as they interact with each other (dry vegetation reflects very differently from moist vegetation, likewise for soils and road surfaces and even roofs of buildings). Thus, the importance of temporal depth, or multitemporal data, to properly capture the reality on the ground as it is differentially affected by seasonality-related factors.

Archived satellite data and historical aerial photography are rich sources of information that can be processed to yield temporally-deep land cover change maps and other products for selected points in time and for the generation of change-images that depict, for example, “from-to” changes of land cover types, and as well as changes in variables such as greenness, leaf area, or plant biomass (Macleod and Congalton, 1998). Aerial photography may extend the satellite image time-series in time, offer alternative spatial resolutions or improved minimum mapping units, and provide the analyst with vertical and three-dimensional perspectives for measurement and interpretation. Also, low- and high-oblique air photos provide coverage of large geographic areas and can be aligned with certain geographic features having specific orientations (e.g., looking at land cover up or down valleys or along political or functional land boundaries). Scan-digitizing individual frames of aerial photography and then correcting and combining them into a seamless image mosaic can effectively transform the remotely sensed data from analog to
digital and prepare the data set for image processing using similar approaches and techniques as used with satellite data.

The temporal depth of the archived data provides historical context and offers “snapshots” in time of landscape characteristics, possibly linked to people through longitudinal surveys, to place through georeferencing procedures, and to the environment through field-based vegetation and soil surveys. Socioeconomic and demographic data collected as part of a longitudinal survey may be retrofitted to relate in time to acquired remotely sensed data by assembling an image time-series in which selected imagery is biased towards survey periods and events that might have occurred prior to the survey, but has implications for the obtained survey results (e.g., droughts, migration events, or changes in road/river accessibility). Place characteristics might involve the geometric correction of the acquired imagery, the transformation of the data from path-row coordinates into UTM (Universal Transverse Mercator) Earth coordinates, and the relative or absolute alignment of multiple images within a time-series. Environment characteristics might involve the use of pattern metrics and a land cover classification in which the spatial organization or structure of land cover types are tracked across an assembled image time-series. Operating at the landscape, class, or patch scales, pattern metrics include such measures as juxtaposition, perforation, fragmentation, and edge characteristics -- useful descriptors of how LCLUC composition and patterns are changing as an indication of landscape form and function (McGarigal and Marks, 1993).

Among the challenges of using an image time-series for capturing compositional and pattern dynamics is how land cover will be categorized for historical periods. Classification approaches (a) use training data to compare areas of known cover-types to unknown spectral responses through supervised approaches, (b) search for “naturally” occurring spectral responses and their possible convergence into spectral clusters through unsupervised approaches, or (c) integrate the supervised and the unsupervised approaches through hybrid schemes. In the supervised approach, field data, aerial photography, and/or other forms of land cover “control” are needed to define the location and composition of “training” areas, whereas in the unsupervised approach, cluster labeling of spectral patterns as to land cover type generally relies upon statistical measures of spectral association and separability as well as field or imagery data to give landscape meaning to the clustering statistics and the defined spectral classes. The hybrid approach is often used as a way of overcoming the limitations of each approach, and taking advantages of the strengths of the first two approaches (Mausel et al., 1993; Moran et al., 1994; Moran and Brondizio, 1998).

Characterizing landscape state and conditions variables for historical periods often increases the complexity of the classification process, because field data frequently are not available and/or aerial photography may be absent. Therefore, remote sensing analysts often opt for generalized classification schemes to minimize errors of commission and omission, rely upon relative accuracies and not measures of absolute classification accuracy, or search for alternative ways for validating historical data sets using ecological and/or demographic techniques (though very rarely done). Such approaches might use correlated ecological data, retrospective survey data to construct landscape chronologies, or the construction of “panel” data sets so that land use histories can be derived and illogical land cover pixel-derived trajectories, such as forest-to-water-to-forest occurring within a 3-year period, can be interpreted as classification error, because the change magnitudes and directions are believed to be improbable, given local site conditions and hypotheses about land cover dynamics.
Even for contemporary remotely sensed imagery, the spatial resolution of the data affects the researcher’s ability to validate land cover classifications. For instance, normal procedures generally involve random, stratified random, or systematic samples arrayed across a classified landscape and often weighted by percent area of each class. Using high resolution imagery as a field guide or a GPS to navigate to preselected and encoded points, quadrant or line transects are often used to characterize the composition and pattern within pixels. Using Landsat TM data, having a 30 x 30 m pixel, for its optical channels, both field survey approaches have obtained acceptable results. But for imagery of large spatial resolutions such as the 1.1 x 1.1 km pixels of NOAA’s AVHRR (Advanced Very High Resolution Radiometer) or the 250 x 250 m and 500 x 500 m pixels of MODIS, more complex field protocols may be needed because of the grain size of the pixel.

Assuming that historical reconstructions of land-cover are robust, major errors may follow if the study of the current socioeconomic conditions are “back-casted”, assuming stationarity in the driving forces of change. As Kelpeis and Turner (2001) demonstrate, the kind, source, and rates of deforestation in the southern Yucatán have changed during modern history in relationship to the vision that the Mexican government has held of it: a dismissed wildland, an extractive forest frontier, an agricultural development zone, and an archaeo-ecotouristic region.

Over shorter time periods, say a decade or two, the issue of land user recall looms large. The basic question is the extent to which people can remember how they used the land during various time periods. To the best of our knowledge, there has been relatively little work examining this issue (though there is a considerable literature in the social sciences on informant recall). The more general issue of how well people recall various things has yielded the conventional wisdom that respondents can recall events like marriages, births or migrations for 20 or more years in the past, but they cannot accurately recall their prior attitudes. This suggests that we cannot simply assume that people can recall land use many years in the past—though they may have a very good idea of what they did this year and even last year. In some unpublished work while designing a questionnaire for Nang Rong Thailand, Walsh, Entwisle and Rindfuss found that people reported having trouble recalling how they used their land parcels more than one or two years in the past. This was based on respondents’ perceptions, and there was no attempt to use outside data to see if the respondent’s perceptions seemed accurate. Moran and colleagues routinely use time-series aerial photos and Landsat TM images for a particular parcel to assist the informants in reconstructing their land use history (McCracken et al., 1999, 2002; Moran et al., 2001, 2002; Brondizio et al., 2002). They find that this method is helpful at the parcel level. It is possible to have an approximate idea of the land cover for a series of time periods, and ask the respondent about land use and cover at these time points. For example, the interviewer can point out that in a particular part of the parcel in 1985 there is an area that is forest-like in appearance and that it is likely either cocoa or 10-year secondary succession. The respondent can then answer which it was. Or, to continue the example, the interviewer can point out that in another part of the parcel an area was being intensively cultivated with some open-field crop. Again, the farmer may be able to resolve this question by remembering the location of the patch and his strategies during that period, say corn or cassava (since they have different soil fertility requirements).

For relatively long time periods, mapping land cover change is further exacerbated by the possible non-stationarity of the landscape and the nature of the spatial autocorrelation of the mapped cover types. If, for instance, the landscape is becoming...
increasingly wet, because of an upward trend in the precipitation levels, finding coarse changes in plant biomass may be biased towards longer analysis periods, when dry and wet conditions occur over anniversary dates, thereby affecting the greenness levels recorded in the imagery. And, the ordering of values as a consequence of location (spatial autocorrelation) might impune accuracy assessments for clustered samples and bias results for change-detections for image dates within the same season or for periods when change is unlikely to occur because of the periodicity of land cover change.

5 Multi-Disciplinary Issues

To engage in multi-disciplinary science means traversing the theories, practices, languages, orientations, perspectives and histories of sometime allied fields but also of distant sciences. However difficult the journey, the research questions that engage the land cover/use research community require the integration of social, natural, and remote sensing-GIS sciences and their perspectives. Multi-disciplinary teams are often assembled to accommodate this diversity of expertise by including scientists that have training and experiences in one or more fields of study so that collectively, broad swaths of the social, natural, and spatial sciences are represented. While the nature of the multi-disciplinary team may evolve over time to address new questions and to take on new research opportunities, core strengths in the human-environment-spatial domains are fundamental. Because of the relevance of local culture and context, a regional specialist, often a collaborator from an in-country or local institution also participates on the team as a core area of emphasis. Further, technical expertise is critical and so statistical programmers, spatial programmers, spatial analysts, and survey specialists are among the support group that is critical to the success of the research venture and to the overall performance of the team. Beyond research, members of university-based teams also have training responsibilities that may include developing in-country capacities, developing spatial and demographic survey teams for data collection, and instructing students about research methodologies and protocols, including research ethics and issues of data confidentiality.

A framework that integrates, for instance, soils, climate, and hydrology as descriptors of the environment with human causes and consequences, represented and integrated within a spatially-explicit context afforded through remote sensing and geographic information systems, supports many of the cross-cutting issues that seek to integrate people, place, and environment. To support such a framework, scholars impose theories and practices from their disciplines to help form questions and frame hypotheses, and often embed disciplinary practices into the analytical design. Among those topics are questions of the directionality of relating people to the environment. Do we begin with people and link to the land, or do we begin with the land and link to people? Generally, spatial, landscape and natural scientists may wish to initiate research by framing questions that begin with the land, represent landscape characteristics as continuous surfaces, and reference features within Earth coordinate systems. Often, these scientists may see the environment or the spatial-temporal pattern of land cover as the dependent variable. Social scientists may wish to start with people and represent them at discrete locations, possibly referenced through areal units of aggregation. Often too, social scientists might seek to explain a socioeconomic and/or demographic characteristic of a household or community, transforming continuous geographic and/or biophysical variables into discrete values for
compatibility. As discussed earlier, the starting point of the analyses—land or people—creates different selectivities that need to be handled through different analytical designs.

Data quality is important to land cover and use characterization that is subject to debate and disciplinary approaches. In the remote sensing community, land cover is normally assessed through an error matrix that compares the remotely-sensed classification to the same classes mapped through a data source of presumed higher accuracy. Standard protocols call for the generation of class accuracies that reflect omission and commission errors and errors to the overall classification as a consequence of chance (i.e., the kappa statistic, Hudson and Ramm, 1987; Congalton 1991). An interpretation of off-diagonal errors is used to assess the nature of the confusion between classes and to possibly consider the merging of classes to minimize misclassification. Also, remote sensing specialists often use non-spectral information such as digital elevation models in the classification process either as an additional classification vector operating at the pixel level or in a post-classification stratification where topographic settings separate spectral classes based upon location and landscape strata. Non-spectral data are used to increase the quality of the land cover map. But by including the non-spectral, ancillary data, endogeneity may result when the derived land cover classifications are used in an assessment or modeling activity in which terrain is included or a variable such as soil moisture potential is used that may be highly correlated with topography.

Classification accuracies of Level-1 (e.g., forest, agriculture, urban) mapping are generally around 90 percent, but accuracies in the 60-80 percent levels are not uncommon for more detailed classifications (e.g., deciduous forest, cropland, residential). Also in an attempt to improve classification accuracy of land cover types, multiple time periods may be consolidated into the same classification by clustering pixels through multiple spectral space, beyond the spectral channels of a single image scene. Often, spectral derivatives in the classification process such as greenness measures derived through vegetation indices are used. Finally, it is relatively uncommon for accuracy reports to contain a full error audit including pre- and post-processing, or statistics that report the spatial structure of classification error (Brown et al., 2000).

Answering land cover and land use change questions requires the integration of the social, natural, and spatial sciences. This, in turn, creates its own set of challenges. Different disciplines have different “defaults” in terms of what questions are interesting to investigate and the best way to address them. Starting point—land or people—is certainly an example. Whereas those trained in the social sciences will tend to start with people, those trained in the natural and spatial sciences will tend to start with the land. A related difference is that those trained in the social sciences will tend to prefer a relatively micro approach, focusing on people and households where the science and data are strongest. Relationships between biophysical, spatial, and social variables are scale dependent (Walsh et al. 1999). Something seemingly simple such as a match rate between households and field plots can be measured in different ways, and the different disciplines may have differing perspectives on which one is “right.” In the case of a match rate, the value will depend on how one sees the task, the conceptualization of the population at risk, and the definition of the denominator in terms of people or land. Prevailing norms about “best practice” also vary between disciplines. For example, the statistical modeling of causal influences and the proper treatment of potential endogeneity is a major issue right now in the social sciences, especially the economic and demographic sciences. Multi-disciplinary teams are well positioned to make progress at the interface of the social, natural, and spatial
sciences, but the challenges are large and at the same time, sometimes subtle and difficult to see.

6 Next Challenges

We conclude by returning to a theme from the introduction. The history of linking pixels and people is short, and the research community will continue to develop methodological and technical solutions. We conclude by discussing some research areas that could involve the linking of pixels and people, areas that are potentially important land use change research topics, and yet thus far have received limited research attention. We expect that part of the reason for this involves the difficult pixels and people linkages that might be required.

Most of the projects that have so far linked people and pixels at the micro level have done so in rural areas, primarily in developing countries, and quite frequently in frontier or recently frontier areas. Conversion of forest into agricultural use has been a principal focus, and typically the agriculture involved growing crops for subsistence use. Commercial crops have tended to receive less attention, but there are examples such as cassava in Thailand (Rindfuss et al. 2003b) and chilis in Mexico (Turner and Geoghegan, 2003). Progress has been made in devising ways to link people and pixels in rural areas of developing counties, but, as we have noted repeatedly throughout this paper, there are many methodological issues that require further work before the field should feel secure in its approaches. We will not repeat those issues here, but rather point to some challenges that are likely to arise as the land science community links more diverse groups of people and pixels.

Once one moves beyond subsistence agriculture and similar types of land use (hunting and gathering, pastoral activities, or wood collection for cooking) to land use that explicitly involves an exchange of money or other goods, then there are two groups people (individuals, households, and various corporate groups) involved. For linguistic ease, we will use the “producers and consumers” terminology, aware that for some purposes the connotations break down. Axinn and Barber (2003) refer to the distinction between direct and indirect consumption of environmental resources that emerges with specialization, industrialization, urbanization, and the expansion of markets from local to regional to global.

Above we gave the example of vacationers affecting land use in interesting but fragile settings. There are numerous other potential examples. A Midwest U.S. farmer growing corn that might be consumed in a variety of countries around the world, or fed to cattle, which in turn could be consumed in an assortment of places outside the Midwest U.S. Most shrimp consumed in the U.S. is farm-raised in Latin America and Asia. Affluent Europeans might have their kitchen cabinets built from koa wood from Hawaii, mahogany from Vietnam, or zebra wood from central Africa. As world demand for coffee increases, farmers in Kenya or Brazil might react by planting more coffee, and if the medical community ever has definitive proof that drinking coffee injures one’s health, land use in the world’s tropical and subtropical areas would change appreciably.

The general point is straightforward. As consumers within countries and globally make decisions about what to buy, this affects how producers make land use decisions. Sometimes the consumer-land use change linking relations are concentrated and clearly
evident: a religious organization (or a corporation) is building a new church (a new corporate headquarters) that requires a substantial amount of ebony and a hectare of forest is deforested (harvested) in Mauritius. (See Lutz and Holn 1993 for a discussion of world market demands and deforestation in Mauritius.) Here one could begin to imagine a research design that linked the consumption decision to the land use decision to the pixel classification change. Sometimes the consumer-land use change linking relations are very diffuse: a change in tastes such that consumers in affluent countries are willing to pay more for produce that is locally grown, organic, and freshly picked. While it seems self evident that diffuse changes in consumer preferences and purchasing behavior will produce diffuse land use changes, we know of no studies that have linked diffuse groups of consumers to changes in land use as measured in pixel classification change. We would hypothesize that as markets become more global and interdependent, there will be a tendency for land use decision makers to respond to global markets and use their land to maximize profits rather than for sustenance. Given that these land use changes may have detrimental effects globally (deforestation is a good example) or locally (the environmental problems associated with shrimp and prawn farms), it will be important to go beyond “self evident” understanding and examine the pathways that link diffuse change in consumer behavior to change in land use.

Another challenge will be to bring the lessons learned in linking people to pixels in rural areas to urban areas. Given that 47 percent of the world’s population already lives in urban areas, and that percentage is projected to grow to 60 percent by 2030 (http://www.un.org/esa/population/publications/wup2001/WUP2001report.htm), improved understanding of urban land use change is important. So far studies linking social science data with remotely sensed data for urban areas have done the linkages at fairly macro levels, such as district or county (e.g. Seto and Kaufmann, 2003). While these studies have yielded interesting findings, it will also be important to link urban land use decision makers to the parcels and pixels over which they exercise decision making authority.

7 References


BurnSilver, Shauna B., Randall B. Boone, and Kathleen A. Galvin. 2003. “Linking Pastoralists to a Heterogeneous Landscape: The Case of Four Maasai Group Ranches in Kajiado District, Kenya.” In

---

5The history of science is littered with examples of self evident understandings that later proved not to be the case. “The world is flat” is perhaps the best one.


CHAPTER 23

MODELING LAND USE AND LAND COVER CHANGE

DANIEL G. BROWN¹, ROBERT WALKER², STEVEN MANSON³, KAREN SETO⁴

¹School of Natural Resources and Environment, University of Michigan, Ann Arbor, MI 48109 USA, danbrown@umich.edu
²Department of Geography, Michigan State University, East Lansing, MI 48824
³Department of Geography, University of Minnesota, Minneapolis, MN 55455
⁴Institute for International Studies, Stanford University, Stanford, CA 94305

1 Introduction

Models are used in a variety of fields, including land change science, to better understand the dynamics of systems, to develop hypotheses that can be tested empirically, and to make predictions and/or evaluate scenarios for use in assessment activities. Modeling is an important component of each of the three foci outlined in the science plan of the Land use and cover change (LUCC) project (Turner et al., 1995) of the International Geosphere-Biosphere Program (IGBP) and the International Human Dimensions Program (IHDP). In Focus 1, on comparative land use dynamics, models are used to help improve our understanding of the dynamics of land use that arise from human decision-making at all levels, households to nations. These models are supported by surveys and interviews of decision makers. Focus 2 emphasizes development of empirical diagnostic models based on aerial and satellite observations of spatial and temporal land cover dynamics. Finally, Focus 3 focuses specifically on the development of models of land use and cover change (LUCC) that can be used for prediction and scenario generation in the context of integrative assessments of global change.

Given space limitations, we focus on spatially explicit models of LUCC. Because the majority of models of this sort are implemented at relatively local scales - sometimes called landscape scales (e.g., 1-100,000km²), we focus on these scales. These models, therefore, may not be appropriate for scaling up to continental and global scales. However, we discuss needs and prospects for models coupling cross-scale dynamics towards the end of the chapter.

In the next section, we summarize existing literature reviews of LUCC models, with a focus on typologies of models. Next, we discuss the importance of and approaches to addressing causation in LUCC processes. We then discuss specific modeling approaches in two broad categories: empirically fitted models and process simulation models. Though there is a good deal of overlap between these two categories, the categories address a fundamental distinction in the ways models have been built. Empirically fitted models are inductive, in that they seek descriptions of processes based on data measuring outcomes from those processes observed at specific places and times. We present an illustration from a NASA LCLUC project in China. Process simulation models are generally deductive, in that they start with a general understanding about processes and seek to build models that simulate outcomes in
specific places. In this context, we discuss the substantial challenges of calibration and validation and present an illustration from a NASA LCLUC project in Mexico. We conclude with remarks about the needs and prospects in LUCC modeling.

2 Types of LUCC Models

Literally hundreds of models of LUCC have been described in the literature on landscape ecology, geography, urban planning, economics, regional science, computer science, statistics, geographic information science, and other fields. Because of differing disciplinary perspectives, as well as differing methodological approaches, data availabilities, and modeling goals, attempts to categorize models are complicated by a relatively large number of dimensions on which the models vary. A number of reviews of LUCC models have been produced in recent years, each from their own perspective and producing a number of different typologies. What follows is a brief summary of several of these efforts, and an outline of the categories of models discussed in this chapter.

Perhaps the first of these reviews was published by Baker (1989) in the context of landscape ecology, so its focus was on land cover change. Models were grouped according to the goals of the models. Whole landscape models seek to model change in some aggregate attribute or state of the landscape over time. Distributional models describe changes in the proportion of the landscape in each of a number of land cover classes. Spatial landscape models describe the location and configuration of changes in land cover. Focused as it was on landscape ecological processes, the paper by Baker (1989) did not discuss models that included explicit representation of human decision-making.

A pair of publications in the late 1990s reviewed the significant amount of work that had gone into modeling tropical deforestation (Lambin 1997; Kaimowitz and Angelsen, 1998). Whereas the review by Lambin (1997) described models of observed land cover change that used mathematical, empirical/statistical, and spatial simulation models, Kaimowitz and Angelsen (1998) focused on land use change models that were developed to describe micro-, regional-, or macro-economic aspects of the deforestation process, using similar methodological categories. Later, Irwin and Geoghegan (2001) compared and contrasted non-economic models, which included many of the approaches described by Baker (1989) and Lambin (1997) but also included cellular automata (CA), and economic models of LUCC, which were divided into non-spatially explicit and spatial explicit models.

Another recent review offers an alternative typology of land use change models. Agarwal et al. (2002) described 19 models that were arrayed on dimensions of space, time, and human decision-making. Models were characterized on these three dimensions according to both the scale at which they operated and the degree of complexity in the representation. The attention to the ways in which human decision-making is represented, in both the reviews by Kaimowitz and Angelsen (1998) and Agarwal et al. (2002), draws attention to the need for models that represent human decision-making explicitly. Another recent review (Parker et al., 2003) focused on agent-based models as tools for representing human decision-making and simulating the aggregate outcomes that result from decisions made by many individuals. Agent-based models represent a qualitatively different approach to the mathematical and
statistical approaches that pervaded the earlier reviews, and offer potential for new LUCC models, similar to the approach to ecological modeling offered by individual-based models (DeAngelis and Gross 1992). Parker et al. (2003) argue that combining agent-based models, to represent human level decision-making, with cellular models, to represent biophysical landscape change, offers a promising approach for future model development in LUCC.

An important distinction that is made in only a few of these reviews is that between models of land use change and models of land cover change. While the reviews by Kaimowitz and Angelsen (1998), Irwin and Geoghegan (2001), and Agarwal et al. (2002) explicitly focused on models of land use change, those of Baker (1989) and Lambin (1997) are oriented towards models of land cover change. The review by Parker et al. (2003) appears to be the first to address both land use and land cover change, though it has a specific focus on agent-based models. The distinction is important because it affects both the data requirements for calibration and validation and the process representations required. There is not always a one-to-one relationship between the two (Cihlar and Jansen, 2001) and, while human activity defines land use, land cover change can proceed with or without a proximal human driver (e.g., through climate change). If we are to bridge human activity and ecological structure and function, representations of both processes are required, adding greater dimensionality to considerations of models in this field.

For this chapter, we describe two groups of models according to whether the modeling approach is oriented primarily toward (a) fitting data or (b) simulating processes. The first category of models includes a broad range of models that employ social science theory and represent decision-making and biophysical processes to varying degrees. We start by describing a range of fitted models of LUCC that help us understand how much LUCC, and of what types, is occurring where. Because of the large number of studies that have involved land use models within an econometric framework, and because of the readier availability of land cover data from remote sensing, we distinguish between fitted land cover models and econometric land use models. Next we describe process simulation models of LUCC that are built as generative representations of the elemental processes of agent decision-making and/or biophysical landscape change for simulating change outcomes.

### 3 Causation in LUCC Models

LUCC are responses to social and ecological processes on a landscape. Much has been written about the causes of LUCC, both in specific contexts, like the tropics, and in general (Fujita 1989; Fujita et al., 1999). Because of the complexity of causes of LUCC, a useful and commonly held distinction is between the *proximate* and *ultimate* causes (Geist and Lambin, 2001; 2002).

Causal explanation draws on both the social and natural sciences. In the social sciences focus is on the land manager, or the so-called "agent." Agents of interest are often those engaged in agricultural activity (e.g., Lynam 2003) or residential development (e.g., Bockstael 1996). Anthropologists, economists, geographers, and sociologists typically take the agent as the point of departure for analysis, reaching out to ever broader contexts to identify the circumstances and conditions affecting the behavior and social processes of interest. Political ecologists (e.g., Blaikie and
Brookfield, 1989) and econometricians (e.g., Chomitz and Gray, 1996), for example, tend to prefer explanations in which individuals pursue behavioral objectives (e.g., profit maximization, risk minimization) within the constraints of their social and ecological circumstances, and in response to broad-scale forces. Ecologists have focused on the ecological and edaphic constraints on human activity, such as when they affect decisions about placement of nature reserves and agricultural activities (e.g., Huston 1993). Further, ecological processes associated with disturbance, species range expansion, and competition can result in land cover changes, either independently or in interaction with human activity. The proximate human causes of LUCC, namely the actions taken by individuals, are therefore promoted and constrained by underlying factors, such as biophysical variability and feedbacks, social structures and macro-economic circumstances (Geist and Lambin, 2001; 2002).

Types of LUCC differ depending on the types of agents that are active on the landscape. In a large and/or heterogeneous region, different agents may be active in different areas, which adds spatial complexity to any attempt at causal explanation (Walker et al., 2000). Further complicating the matter is that LUCC may be substantially affected by dynamic interactions, among agents or between agents and their environment (Geist and Lambin, 2001). The deforestation literature has long recognized interactions between loggers and follow-on farmers, for example (Walker 1987; Rudel 2002). Further, there is increasing evidence that negative feedbacks associated with prior development play an important role in urban land development decisions (Irwin and Bockstael, 2002).

The agents and their interactions have been mainly described as constituting the proximate causes of LUCC (Geist and Lambin, 2001; 2002), in which case it is necessary to consider the ultimate forces in order to arrive at an encompassing account of causality. While research on the ultimate causes has tended to emphasize so-called “economic” factors (e.g., Kaimowitz and Angelsen, 1998; Irwin and Geoghegan, 2001), these are shaped by underlying environmental heterogeneity and variability, demographic change, technological evolution, and institutional intervention (Walker et al., 2002). Thus, there are two primary scales of processes at work in LUCC, a micro-scale, in which individuals seek to achieve their objectives, and a macro-scale, reflecting the context of these decisions. The context of decisions include such processes as population growth and movement, the climate and soil processes that constrain production on the land, the evolution of commodity and labor markets, the continuing progress of technological change, and the actions of government bureaucrats responding to political forces. Changes in the macro-scale are often unpredictable, given that they result from complex interactions of economic, political, and transnational institutions and social processes. It is these macro-scale processes and changes that are usually interpreted as ultimate causes in LUCC research.

Finally, it would be a mistake to conclude that there is a one-way causal path, running from macro-scale forces to the micro-responses of individuals. Indeed, individuals and communities show remarkable resilience and ingenuity in adapting to a changing context to pursue their own objectives. Thus, while macro-scale factors no doubt shape the realm of choices available to agents, they by no means predetermine any particular land use or cover outcome.
4 Empirically Fitted Models

The first type of models that we describe includes those that are empirically fitted, i.e., they are based on statistically matching the temporal trends (in the case of longitudinal data) and/or spatial patterns (in the case of cross-sectional data) with some set of predictor variables. The variables to be predicted may represent land use, land cover, change in either of these or some combination and may be measured over pixels, derived from remotely sensed data, parcels (i.e., irregularly shaped spatial units defining legal ownership) or aggregated over some jurisdictional unit. Predictor variables are factors thought to influence land use, including proximity to roads, proximity to cities and towns, mix of economic activity, demography, income and wealth, and biophysical factors like slope and soils.

Some of the earliest empirically fitted models represented LUCC as Markov random processes, in which the state of the land at some time in the future was a function only of its present state, represented by a transition probability (Burnham 1973; Bell 1974; Muller and Middleton, 1994). Though these simple models have appeal for their minimal data requirements and analytical properties, their assumptions are restrictive and they have little utility for analyzing policy. To address some of the restrictions, later models have used the Markov framework, but added a dependence on neighboring states (e.g., Turner 1987) and introduced spatial and temporal non-stationarity to the transition probabilities (Sklar and Costanza, 1991; Brown et al., 2000). Future applications of Markov modeling to LUCC will need to incorporate these modifications, which means the models are not strictly Markovian, but rather a more general form of stochastic model.

Statistical estimation methods are more commonly used to fit LUCC models. Some of these models have been constructed with only loose connection to theory, while others are constructed in a more rigorous, usually economic, theoretical context. Given the discrete nature of land use and cover categories and changes, the most common approach is to estimate a logistic regression function that describes either the probability of a particular land category occurring or of the location transitioning from one land category to another. This approach has been used for modeling development in urban areas (Landis 1994; Landis and Zhang, 1998a; 1998b), along urban-rural gradients (Wear and Bolstad, 1998), and in tropical forests (Chomitz and Gray, 1996; Mertens and Lambin, 1997).

A number of limitations challenge the utility of simple statistical models, and solutions have been developed or are in development to deal with these challenges.

- The assumption of log-linear relationships between the predictor variables and the dependent variable can be restrictive. In order to relax this assumption, a number of non-linear fitting methods have been used to describe land use or cover change, including generalized additive modeling (Brown et al., 2002), which is a non-linear statistical method (Hastie and Tibshirani, 1990), and artificial neural networks (Pijanowski et al., 2002), which is a networked learning approach that comes from artificial intelligence research.

- The assumption of temporal stationarity (i.e., that the model parameters remain constant over time) leads to models with misleading R^2 values and t-statistics. These problems can be dealt with using econometric methods developed for analysis of panel or time series data (e.g., Hsiao 1986). An illustration of a panel data analysis is presented below.
Spatial autocorrelation and non-stationarity is also likely to affect estimation efforts because of the effects of contiguity in spatial data, together with the diffusionary nature of many land change processes (Walker and Solecki, 1999; Walker et al., 2000). It is necessary to account for the influence of spatial autocorrelation in these models because, otherwise untreated, it can lead to erroneous conclusions about the relationships between dependent and independent variables. Wear and Bolstad (1998) included a neighbor-weighted dependent variable in the list of predictor variables, whereas Bell and Bockstael (1997) used a method that incorporates the spatial weight matrix in a generalized-moments estimation. The method presented by Bell and Bockstael (1997) was applied to household level data, but could be extended to pixel- or parcel-level land use and cover data.

Failure to incorporate information (often survey-based) about the household or community structures can create specification bias (Walker et al., 2002), because land use processes may be different for different types of households or communities.

Empirical models can suffer from the ecological fallacy – when the characteristics of an individual agent are inferred, often incorrectly, from estimation based on aggregated observational units – and the modifiable areal unit problem – where the shape and size of data aggregation affects analysis (Chou 1993). Variables differentially co-vary as a function of the scale at which they are measured; scale can therefore act as an independent variable (Bian 1997; Walsh et al., 1999) and affect the apparent magnitude and direction of relationships among causal factors (Kummer and Sham, 1994).

Many empirical models assume unidirectional causal change, i.e., income or population growth causes deforestation or urban growth, but that there are no bi-directional feedbacks or testing for causality. Endogenous interactions and feedbacks have been incorporated into some models (e.g., Chomitz and Gray, 1996), but many empirical models assume them away. Models that fail to identify the endogeneity of variables will be misspecified and policy recommendations may be incorrect.

4.2 PANEL APPROACHES IN THE PEARL RIVER DELTA

Panel analysis methods represent promising options for developing empirically fitted LUCC models, as this example demonstrates. Among the chief advantages to panel approaches is the ability to allow the relation between the independent variables (drivers of land use change) and the dependent variable (land use change) to vary across time and space (Hsiao 1986). Panel econometric techniques were used to model land use change in the Pearl River (Zhuijiang) Delta (PRD) in South China (Seto and Kaufmann, 2003). Two important aspects of this study affected the choice of modeling methodology employed. First, the remote sensing data used to extract land use trajectories consisted of nine consecutive images from 1988 to 1996. Although this time series is longer than many land use change studies, it is a relatively short period that does not allow sufficient degrees of freedom to estimate a statistically reliable model if the entire study area is treated as a whole. Thus, to increase the reliability of the statistical results, the land use data were augmented by cross-sectional data at the
smallest possible administrative unit, the county. Second, the main objective of the study was to evaluate the macro-level causes of land use change, and in particular, the urbanization of agricultural land and natural ecosystems. Because the aim was to understand the relation between policy reforms and land use, a modeling approach appropriate to address this level of analysis was required.

As with all fitting techniques, the specification of the model is an important determinant of the quality of the results. Model misspecification can lead to spurious results. This is especially the case with panel data, where model coefficients can vary both temporally and spatially. In the PRD case study, panel statistical tests were used to identify the appropriate model based on the characteristics of the data (see Hsiao 1986). A result of this analysis was that the socioeconomic factors correlated with land use change were shown to vary by county. This indicated that a random coefficient model was most suitable. Following estimation of the models, causation among the dependent and independent variables was explored using the method of Granger causality (Granger 1969; Granger and Huang, 1997), which examines lagged correlations in time-series data to identify chains of causation. Results from the analysis of urbanization of natural ecosystems revealed complex interactions and feedbacks among the various factors (Table 1). Two of the independent variables, investment in capital construction and relative rates of labor productivity, appeared to ‘Granger cause’ the conversion of natural ecosystems to urban uses, i.e., they were correlated with the conversion in space and preceded it in time. This type of land use change also appears to have a feedback on capital construction, suggesting that high rates of urbanization attract additional investments in construction. These results suggested that land use change occurred on a nested-hierarchy of scales, and that the underlying driving forces were not simple, nor necessarily unidirectional. The techniques used in the PRD illustrate some of the ways in which advances in theoretical and applied econometrics are being adapted for land use change modeling.

Table 1. Analysis of causal order for the natural ecosystem→urban model (Seto and Kaufmann, 2003).

<table>
<thead>
<tr>
<th>Causal Variable</th>
<th>natural→urban</th>
<th>Capital Investment</th>
<th>Ag/Urb Land Productivity</th>
<th>Ag/Urb Labor Productivity</th>
</tr>
</thead>
<tbody>
<tr>
<td>natural→urban</td>
<td>-</td>
<td>32</td>
<td>9</td>
<td>10</td>
</tr>
<tr>
<td>Capital Investment</td>
<td>21</td>
<td>-</td>
<td>1</td>
<td>6</td>
</tr>
<tr>
<td>Ag/Urb Land Productivity</td>
<td>8</td>
<td>14</td>
<td>-</td>
<td>23</td>
</tr>
<tr>
<td>Ag/Urb Labor Productivity</td>
<td>21</td>
<td>17</td>
<td>14</td>
<td>-</td>
</tr>
</tbody>
</table>

Cell values refer to the number of occurrences, out of 165 sub-samples, where the causal variable “Granger caused” the dependent variable. Values in bold are statistically significant at p< 0.05.
5 Dynamic Process Models

In contrast to empirically fitted models, dynamic process models of LUCC seek to represent the most important interactions between agents, organisms, and their environment as computer code. Though fitted models can be used to generate simulations (e.g., Brown et al., 2002), simulation is clearly more central to the use of process models. Although these models need to be calibrated and validated, they deemphasize the fitting of data and emphasize the fidelity of model elements and processes to what is known about the processes. Here we describe three types of dynamic process models that have been used for LUCC: process flow models, cellular automata (CA), and agent-based models.

Models that track material and energy flows through a landscape and how they are transformed and/or transmitted at each location (e.g., Fitz et al., 1996), work well for some natural processes and to represent stable, stylized socioeconomic systems. Models of this sort have been used in a variety of settings, including the Everglades (Wu et al., 1996) and the Patuxent Watershed (Voinov et al., 1999). These modeling approaches often include enough stochasticity that they become simple simulation platforms to evaluate empirical models as described above. Their primary limitation for modeling LUCC, however, is an inability to represent the complex ecological or social interactions, including competition and cooperation, the role of institutions, interest groups, and limits to rational behavior, that give rise to behaviors and decisions that affect land use and cover.

Cellular automata (CA) and other cellular models are fairly common in the land use modeling literature (e.g., Batty and Xie, 1994; Batty et al., 1997; Clarke and Gaydos, 1998). In some CA models, the cells are considered simple actors with fixed neighborhood relations and update rules. In other models, the CA represents the state and dynamics of the environment. The cells can represent parcels of land, each with their own characteristics and each changing as a result of some fixed rules based on the cell's state and the state of its neighbors. CA models, coupled with data on the heterogeneity of the environment, can capture important dynamics of LUCC, including diffusive, road-driven, and spatially random changes (Clarke and Gaydos, 1998). Further, because they are dynamic and iterative, CA models can represent endogenous interactions and feedbacks. An important challenge in the development of CA models is how to establish the rules that govern system behavior, and incorporation of heterogeneity and dynamism into these rules. Also, the simplified decision rules of CA models make policy interventions difficult to explore at the level of individual decision makers, social groups, or institutions.

Agent-based models (ABMs) are defined in terms of entities and dynamics at a micro-level, i.e., at the level of individual actors and their interactions with each other and with their environment (Epstein and Axtell, 1996; Kohler et al., 2000; Gimblett 2002; Janssen 2003a; Parker et al., 2003). ABMs consist of one or more types of agents, as well as an environment in which the agents are embedded. Agents in ABMs may be individuals (e.g., householders, farmers, developers) or institutions (e.g., townships, NGOs, firms; Bousquet et al., 1998; Gimblett 2002; Janssen 2003b). Therefore, systems can be studied at many scales and parts (specified at different scales) can be integrated into a coherent whole. Agent specification requires defining their state (e.g., preferences, memory of events, and social connections) and their decision-making rules, heuristics and other mechanisms to perform particular
behaviors. The agents generate their individual behaviors in response to inputs from other agents and from the environment. Agents may also adapt, or change their behavior, through learning or evolution based on their experience. Adaptive behaviors are usually represented in the models using some sort of learning algorithm, such as genetic algorithms. As the model is run, agent behavior is generated as agents use their rules to determine which other agents to interact with, what to do when they interact, and how to interact with the environment. Representing human decision-making in ABMs remains an active area of research (e.g., Balmann and Happe, 2002; Berger 2002; Hoffman et al., 2003; Lynam 2003).

The environment of an ABM typically represents the physical environment, e.g., land, water, roads or other infrastructure. The environment at any location has associated states, e.g., land cover type, soil quality, and aesthetic quality. The environmental entities in a model may have their own dynamics, describing how they change in time both independent of and as a result of agent behavior, e.g., to represent soil erosion processes, forest growth and other aspects of environmental change (e.g., Janssen et al., 2003; Lynam 2003). These dynamics are often represented by coupling with CA-based models.

The agents' behaviors affect each other and the environment. The environment changes in response to agent behaviors, but also by following its own dynamics. Thus ABMs embody complex interlaced feedback relationships, leading to the nonlinear, path-dependent dynamics often observed in complex systems. Note that the model's output is both the micro-behavior of agents and the environment, as well as the emergent macro-level structures, relationships and dynamics which result from the micro-level activity. Spatial relationships can be incorporated into agent behavior in order to create more realistic interaction with neighboring agents, such as when a farmer is more likely to learn from a nearby neighbor than from a farmer located far away (e.g., Polhill et al., 2001). However, the models also can include social networks of various kinds, each defining an interaction topology based on, for example, membership in groups, business contacts, and common information sources. These characteristics facilitate the investigation of three particularly important properties that define complex systems: emergence, scaling, and feedbacks. Parker et al. (2001) argued that spatial landscape metrics can be used as measurable map properties that "emerge" from individual-level decision-making and interaction. These measures can serve as scaled outputs from an agent-based model, which has inputs at the level of individual decision-making. Recently developed agent-based models have been used to examine feedbacks between land use change and transportation networks (Batty and Torrens, 2001; Waddell 2002) and environmental characteristics (Rand et al., 2002).

5.1 SOUTHERN YUCATAN PENINSULAR REGION INTEGRATED ASSESSMENT

An agent-based process model was developed to project trends and develop scenarios of tropical deforestation and cultivation in Mexico (Manson 2002). Termed SYRIA (Southern Yucatan Peninsular Region Integrated Assessment), the modeling effort was part of a larger project in the Southern Yucatán Peninsular Region that has drawn on historical analysis, ecological and field studies, and remote sensing (Turner et al., 2001).

SYRIA uses a three-component actor-institution-environment conceptual framework to model decision-making of farming households in the context of
socioeconomic institutions and the biophysical environment. Agents represent households whose characteristics are specified by a survey. The survey was aimed at determining household attributes such as food demand and labor availability. Remotely sensed imagery was used to map the outcomes of past household land use decisions. Agent agricultural location decisions were specified using a boundedly rational utility function that guides a multicriteria evaluation. Agents make decisions that optimize their utility and also adapt to experience through an agent-specific population of genetic programs (cf., Chattoe 1998; Dawid 1999).

Land tenure and market forces, such as returns to commercial agriculture and market accessibility, were included factors driving agent decision-making. Institutions were represented by simple agents with largely pre-scripted behavior. The environment, represented by a cellular model, affects agent decision-making by changing such factors as soil quality and cover type according to rules derived from ecological research (Lawrence and Foster, 2003; Read et al., 2003).

SYPRIA has been used to examine a number of scenarios, including the effects of agricultural commercialization, alternative land tenure policies, and different secondary succession regimes. A suite of quantitative and qualitative validation techniques was employed to test the model and examine scenario outcomes. Results to date indicate that genetic programming holds promise for representing bounded rationality and that institutions and population growth are key to understanding the quantity and patterning of LUCC in the study region (Manson 2002, 2003).

6 Calibration and Validation

There are substantial challenges to integrating empirical observations with models, especially process-based models. Some of these challenges are related to scalar dynamics. Social variables tend to act at fine scales while biophysical variables have greater influence at larger scales (McConnell and Moran, 2001). Scale mismatch can occur when multiple processes operate at different scales, such as when household decision-making occurs at a scale different from pertinent biophysical processes (Fresco and Kroonenberg, 1992). A model may be able to represent different processes at different scales (Veldkamp and Fresco, 1996). Given the challenges of multiple scales of analysis, most integrated datasets are available within local projects or regionally integrated networks, such as the Hindu Kush-Himalaya project, LUCC Miombo, and START/Southeast Asia.

For model validation purposes it is important to clearly identify the objectives of a model. Where accurate predictions are the ultimate goal, measures of the accuracy of spatial outcomes are reasonable for validation. LUCC models are commonly evaluated by comparing predicted outcomes to data using measures of map agreement, like the kappa statistic and the receiver operating characteristic (ROC). Pontius (2002) has presented methods to, in addition, identify, separately, the degree to which a model gets the correct amount of a given land use or cover versus getting their locations correct. Where the goal of a model is to represent the process accurately, however, for example so that policy interventions can be evaluated, validation might require evaluating how well a model reproduces certain critical system properties (e.g., a clustered pattern of change or a cyclical land use dynamic). This requires tests of spatial patterns and temporal dynamics (Turner et al., 1989; Pontius and Schneider, 2001), for
example using pattern and texture metrics (Giles and Trani, 1999).Researchers developing agent-based models have not yet identified a canonical set of approaches to calibration and validation of these models (Benson 1995; Batty and Torrens, 2001; Polhill et al., 2001; Waddell 2002). There remains a need for new measures of fit between the model and data that go beyond spatial matching to focus on variability of outcomes and dynamics.

Another important approach to evaluating process simulation models is sensitivity analysis that explores the relationships between model parameters and the state or time path of variables endogenous to the modeled system (Klepper 1997). Other work examines the effects of interdependency, nonlinear behavior and sensitivity to initial conditions (Miller 1998; Manson 2001). Such tests address issues of error propagation and uncertainty, a topic too often left unconsidered in LUCC modeling (Robinson 1994). Cogent sensitivity, error, and uncertainty handling is essential to vet data and ensure a model behaves in ways that are consistent with relevant data.

7 Prospects

Recent developments and applications of LUCC models have addressed a number of key conceptual and methodological issues that have limited these models in the past. These developments hold promise for future use of these models in addressing the pressing issues of which LUCC are an integral part, e.g., human health and welfare, global change, and ecosystem dynamics and sustainability. Nonetheless, work remains to improve the efficacy of methods for model development, testing, validation, and application. This future work will further improve our ability to evaluate the effects and interactions of LUCC on and with people and their environment.

Research on statistical, econometric, and computational methodologies has improved the ability of fitted models to account for non-linear relationships and endogeneity, spatial dependence and temporal non-stationarity. Routine use of some of these methods will take time and effort, including in the development of accessible software. Future research on fitted models should continue to emphasize developing and applying methods for testing for causality, measuring and incorporating spatial and temporal non-stationarity, modeling across scales, and incorporating endogenous factors and feedbacks. Tests for causality can help, for example, to develop appropriate and effective policy. Regardless of these advances, empirically fitted models will always be limited in their ability to explain and predict by the specific data and contexts within which they are calibrated and fitted. While they are useful for identifying correlations and causal chains within a particular conceptual framework and projecting trends within a relatively narrow range of conditions defined by the data, these models are limited in their ability to generate surprising results or to predict in situations where the proximal and ultimate causes are different from those operating in the empirical situation.

Computational process models have shown great promise for LUCC research. These methods will benefit from further development and testing in a range of LUCC settings and under a variety of scenarios. For example, work is needed to establish appropriate rules to govern the behavior of CA models and to incorporate heterogeneity and dynamism into these rules. Agent-based models are among the newest entries into the LUCC field and work remains to develop and test various alternative approaches to
representing human decision-making. As with other modeling approaches, further improvements in agent-based modeling will come through adoption of methods applied in other fields, especially in the general area of complex systems. Development and common acceptance of a set of methods for calibration and validation of the model processes and their outcomes is needed.

6 References


Bell, E.J. 1974. Markov analysis of land-use change: application of stochastic processes to remotely sensed data. Socioeconomic Planning Sciences, 8(60), 311-316.


Manson, S.M. 2002. Integrated Assessment and Projection of Land-Use and Land-cover change in the Southern Yucatan Peninsular Region of Mexico. Graduate School of Geography. Worcester, Massachusetts, Clark University.


Section V  Synthesis and Lessons: Biophysical Change and Beyond
CHAPTER 24

LAND-USE AND LAND-COVER CHANGE PATHWAYS AND IMPACTS

JOHN F. MUSTARD\textsuperscript{1}, RUTH S. DEFRIES\textsuperscript{2}, TOM FISHER\textsuperscript{3}, EMILIO MORAN\textsuperscript{4}

\textsuperscript{1}Department of Geological Sciences, Brown University, Providence, RI 02906
\textsuperscript{2}Department of Geography and Earth System Science Interdisciplinary Center, University of Maryland, College Park, MD 20742 USA
\textsuperscript{3}Horn Point Laboratory, Center for Environmental Science, University of Maryland, Cambridge, MD 21613
\textsuperscript{4}Department of Anthropology, Indiana University, Bloomington, IN 47408

1 Introduction

Of the challenges facing the Earth over the next century, land use and land cover changes are likely to be the most significant. This anthropogenic process affects many parts of the earth’s system (e.g., climate, hydrology), global biodiversity, and the fundamental sustainability of lands. Various estimates indicate that 50 percent of the ice-free land surface has been affected or modified in some way by human activity (Vitousek et al., 1997), while 10 to 55 percent of the net primary productivity has been captured by human land use activities (Rojstaczer et al., 2001). Over the next century, global population is projected to increase by 50-100\% and it is likely that there will also be an increase in the global standard of living. Thus pressures to further convert or manage “natural” ecosystems for human needs as well as capturing more of the global net primary productivity are also likely to increase.\textsuperscript{1}

Understanding of the patterns of land use and land cover change has increased significantly over the last decade (e.g., Turner 2002a). This has been facilitated in part by increased awareness of the issues and by the large number of focused studies directed to understanding the nature of land-cover and land-use change (LCLUC). These studies have made significant advances in furthering our understanding of the socio-economic drivers of LCLUC, the impacts on natural and human systems, as well as feedbacks between natural and human systems. Given the large number of case studies that have been performed, we now have the opportunity to look broadly at the results of these studies to assess if there are fundamental patterns of land-use and land-cover change that consistently appear regardless of global location, social organization, economic state, etc. Furthermore, we can now assess whether there are persistent impacts of LCLUC that can be identified and related to the overall patterns.

Previous studies have attempted to assess whether there is a common pathway of land cover change, linked to common socio-economic drivers (e.g., Turner et al., 1990; Lambin et al., 2001). A recent study of tropical deforestation sought to assess common drivers from an analysis of the results of 150 case studies (Geist and Lambin, 2001). Here

\footnote{The terms natural and undisturbed are used throughout this chapter in reference to landscapes and environments which are only ephemerally managed and used, even if they were significantly altered by human action in the past.}
we develop a typology of (1) land-cover change (pathways), (2) link them to broad drivers (both land uses and their ultimate causes – policy, economics, social, environmental), and (3) address the major impacts consequences of the land-cover conversions. The typology is derived from examination of case studies results conducted under the NASA Land Use Land Cover Change (LCLUC) program since 1997 (which are summarized in this volume), and where appropriate, the results of studies conducted within the broader community of land-change science.

The search for general principles from case studies is constrained by the limits of the various case studies that inform our analysis (i.e., they are specific to particular places, times). Searching for commonalities from the diverse environments and drivers of land-cover and -use change will necessarily be subject to uncertainty and error. However, case studies are essential for informing large-scale syntheses, and their results must contribute to syntheses describing general principles (Lambin et al., 2001). To a large extent the case studies from which we draw examples are focused on European colonization of western hemisphere regions, reflecting past orientations of the LCLUC program, and thus do not necessarily capture land-change processes in other parts of the world. One of the advantages of western hemisphere emphasis is that the time scale for significant changes in many landscapes is compressed relative to other parts of the globe which may extend over thousands of years. We recognize that the Americas were substantially altered by pre-Columbian societies (e.g. Turner and Butzer, 1992; Denevan 2001). However, this region offers an excellent opportunity to understand the processes and impacts of the massive transformation that have affected this part of the world, particularly over the last 150 years.

The spatio-temporal scale of analysis strongly affects the results. A spatial scale that is too large (e.g., continental) will fail to capture the important interrelationships among processes and therefore lack specificity, while a scale that is too small (e.g. a village) will not encompass a sufficient number of interrelationships to understand the region. Likewise, a short temporal scale may miss past human-environment dynamics that reshaped the very landscape under study. We do not resolve the central issue of scale, but attempt to draw from studies cast at intermediate scales.

It is not possible within the scope of this chapter to put forth all the detailed case study results that have informed this synthesis as has been done elsewhere (Geist and Lambin, 2001). We present here representative case studies from a number of different ecological regions in the Americas, but the reader should recognize that much additional data is presented in the proceeding 23 chapters.

2 Representative Land Cover and Land Use Histories

2.1 TROPICAL DEFORESTATION: THE AMAZON

2.1.1 Scope of the Change
The Amazon region experienced deforestation prior to 1975, but on a small scale. The population collapse of indigenous communities by war and disease following European discovery resulted in a pattern of small communities practicing shifting cultivation and moving their settlements frequently (Meggers 1971; Roosevelt 1989; Beckerman 1991). Assessments using Landsat MSS found less than one percent of the Amazon Basin evidenced deforestation in 1975 (though the resolution of MSS probably hid many areas
that were in secondary succession). Initiated in 1970, Brazil’s Program of National Integration, associated with a major initiative to build roads across the Amazon and to settle land along these roads with colonists, began to change the rates of deforestation. The east-west Transamazon Highway, constructed in less than four years, cut a path from the northeast of Brazil to the frontier with Peru. The north-south Cuiaba-Santarem highway and the Belem-Brasilia highway linked, respectively, the central and eastern parts of the Amazon to the central part of Brazil (Moran 1981).

These roads were catalysts of land cover and land use change in the Amazon. Human settlements were promoted by a series of settlement schemes providing attractive incentives and virtually free land, attracted people who quickly began cutting forest in order to ensure their claims to land (Moran 1976, 1981; N. Smith 1982; Fearnside 1986). For the period up to 1988, Skole and Tucker (1993) were able to document that up to 15 percent of the Brazilian Amazon had been deforested and seriously fragmented—a rate close to 0.5 percent per year. This rate actually hides the real local rates of deforestation. In settlement areas the rates of deforestation were commonly in excess of one percent per year, while vast areas remained out of reach of human occupation by Brazilian society. Percentages, too, tend to hide the scope and magnitude of deforestation in the Amazon: one percent of the Brazilian Amazon is equivalent to 50,000 km$^2$ or an area the size of Belgium. Thus, while the percentage of deforestation is higher in Ecuador and Mexico’s tropical forests, the area being deforested in Brazil is several orders of magnitude larger. Recent updates by EU scientists provide a needed reassessment (Achard et al., 2002).

Rates of deforestation in the Brazilian Amazon reached an initial peak near 1987-88, followed by a notable decline. The drop was not a result, as some thought, of more effective conservation or of a more effective set of policies, and turned out to be temporary. It was, rather, the result of hyperinflation and a serious credit deficit in Brazil. After the introduction of the new currency, and effective control over inflation and exchange rates in 1994, the rate of deforestation surpassed (nearly doubled) the first peak of 1987-88, generating serious concern to policy-makers. This second spike in the rate of deforestation can probably be explained by the suppressed rates of deforestation from 1988 to 1993, and the opportunities that economic stabilization offered. Within two years, deforestation rates settled down to the more common rates of about 0.5 percent for the Basin, although in settlement areas the rates remained considerably higher, i.e. above 1 percent annually (Wood and Skole, 1998; Brondizio et al., 2002; Moran et al., 2002; Lu et al., in press).

2.1.2 Trajectories of Land Change
Land change begins with the clearing of forest through slash-and-burn techniques, commonly followed by the planting of annual crops or the creation of pastures. In some cases, fields are kept in cultivation continuously, but this is rare. Only in areas with alfisols of relatively high fertility with favorable texture are there examples of continuous cultivation for over 25 years with some crop rotations in place (Moran et al., 2002). In most places the low nutrient conditions of oxisols and ultisols, dominant in over 75 percent of the Amazon Basin, present constraints to continuous cultivation without major fertilizer inputs—which remain prohibitively expensive throughout most of the Amazon basin. Without fertilizers, farmers have tended to plant pastures and graze cattle at very low densities as a preferred strategy. Cattle ranching has a long tradition in Latin America and receives favorable treatment by policy makers as a repository of value and a hedge against inflation and uncertain economic cycles. It is the traditional tool for occupying large areas
of the vast frontiers of Latin America with few people and labor scarcity (Walker et al., 2000). Thus, Rondonia (predicted in the 1970’s to become a center for cocoa production) and the Altamira region of Brazil, both of which have patches of high quality soils, are dominated by pasture land (Moran 1988). Less than ten percent of the land area is in crops, with less than four percent in annual or staple crops (e.g., rice, corn, beans, manioc), and the rest in some form of plantation or tree crop (e.g., cocoa, rubber, sugar cane, coffee) (Brondizio et al., 2002). Nevertheless, the typical nature of change is one from undisturbed forest to a landscape cleared for management for cultivation or ranching, with a significant component of secondary regrowth on abandoned land.

Farmers experiment with a variety of strategies. They tend to clear more land than they can manage at the outset, and rates of six percent per year are not unusual when first arriving (McCracken et al. 2002). This rate quickly drops as farmers realize the high cost of managing regrowth through secondary successional dynamics (Moran et al., 1994, 1996, 2001; Steininger 1996; Tucker et al., 1998; Laurence et al., 2001; Mesquita et al., 2001; Zarin et al., 2001, 2002). Those with more favorable biophysical initial conditions and some capital move towards plantations and pasture formation; those with less favorable conditions continue to combine annual crops with modest increments in pastures on lands with exhausted fertility as a way of combating the return of woody species by succession. Over time, those with favorable conditions tend to evolve a balance of crops and pasture, while those with unfavorable soil conditions and poor labor and capital resources tend to concentrate most of their land in pastures.

2.1.3 Forest Conservation Efforts
While legislation in Brazil has sought to protect up to 50 percent of the areas occupied by settlers, raising this figure to 80 percent more recently, there is little enforcement of this legislation even if it were wise to do so. Given poor enforcement and the likely fragmentation of these “back of the property” conservation areas, this legislation seems less than effective as a means of conserving flora and fauna biodiversity. Recent evidence from a study in Rondonia suggests that reserves, including extractive reserves, provide the only effective mechanism for conservation in areas of settlement (Battistella 2001). Reserves in themselves do not ensure conservation, but only where local people maintain a vested interest in protecting the forest for their own economic well-being—as in the extractive reserves in Machadinho, Rondonia—may forests be protected from the pressure for occupation and land clearing.

2.1.4 The Amazon in the Context of Global Tropical Deforestation
Research on causes and driving forces of tropical deforestation reveals that neither single factor causation (e.g., poverty, population growth) nor irreducible complexity adequately explain the dynamics of tropical deforestation (Geist and Lambin, 2001). Deforestation is driven by regional causes, of which the most prominent are economic, institutional, and policy factors which seem to drive agricultural expansion, logging, and infrastructure development (Angelson and Kaimowitz, 1999; Lambin et al., 2001). Logging appears to be a more important driver at the outset of forest clearing in Africa and Asia than in Amazonia, where farming and ranching seem to precede logging activities. In Middle America, selective logging (not clear cutting) has provided road networks ultimately followed by farmers (e.g., Turner et al., 2001). The vastness of the Amazon, and the precariousness of infrastructure have probably mitigated the impact of logging in the
Amazon as a primary driver of land change. Recent work by Cochrane (2000, 2001, 2002) and Nepstad and colleagues (1999, 2000) suggests that loggers are beginning to lead the way in places where some primary road infrastructure has been created.

2.2 FORESTATION: NEW ENGLAND

The environmental history of Massachusetts provides a representative case study for landscape experiencing all major techno-economic phases affecting land use (Foster et al., 1998; Hall et al., 2002). Prior to colonial settlement, the Massachusetts landscape was predominantly forested, though there is evidence for some manipulation of the landscape by native populations (Mulholland 1988; Doolittle 2000). The colonial experience witnessed significant occupational growth, ultimately distributed somewhat evenly across those conditions that could sustain cultivation and/or resource extraction. With the expansion of the nineteenth-century industrial revolution, population concentrated in industrial towns, reducing rural population densities into the middle of the twentieth century, despite a sharp rise in overall population numbers. Following World War II, industrial activity subsided as core manufacturing activities relocated (e.g., textiles to the south) and the regional economy shifted to high technology and service industries.

Land-cover and land-use change in Massachusetts followed these transformations, though not as a simple relationship with population (Figure 1). The initial colonization and movement towards the interior was accompanied by significant forest clearing, the majority of which was pasture.

![Graph showing relationship between forest cover and population in Massachusetts](Figure 1. Relationship between forest cover and population. Massachusetts, USA. See text for explanation.)

By the middle 1800s, the region experienced its greatest proportion of cleared land with only 20 to 40 percent of the land remaining forested. Those forested regions that remained were heavily managed for forest products. With the rise in industrial activity and the opening of the American west for settlement in the middle 1800s, there was a large decrease in rural populations and shift in agriculture to market crops to support the growing populations of the industrial town and cities of the region. Furthermore, higher efficiency agricultural practices in the west coupled with efficient rail transportation made the use of agricultural land in Massachusetts uneconomical for all but the highest value crops. Large-scale agriculture abandonment took place during this period, giving rise to an extensive period of afforestation such that by 1950 the region was 70 to 80 percent forested. The last
50 years has seen a decline and fragmentation of this forest cover associated with urban expansion and suburban/peri-urban development.

The environmental impacts from these enormous changes in land cover can only be broadly framed. Extensive measurements were not made with the exception of forest composition and structure (Forster et al., 1998; Hall et al., 2002). At the height of deforestation, the forest structure of remaining stands was one of relatively youthful, even-aged stands. With afforestation, even-aged stands of early successional species (white and pitch pine, red maple, and birch) became established on abandoned agricultural land. Towards the end of the twentieth century, mature forest structures with long-lived shade tolerant species have become re-established (Hall et al., 2002). Associated with these changes in cover and stand properties have been changes in species composition. Except for the loss of chestnut, most of the changes have been in the relative abundance of species with a decline in the abundance of long-lived species (e.g., beech, sugar maple) and an increase in early successional species (e.g., red maple, poplars, white pine). Introduction of exotic pests and pathogens has probably wrought the most significant change on these forests. Chestnut was once a significant canopy species but is now present only as subcanopy sprouts because of a fungal pathogen introduced early in the 20th century (Paillet, 2002). Beech bark disease and hemlock wooly adelgid are additional examples of exotic species causing changes in forest structure and composition (Twery and Patterson, 1984; Orwig et al., 2002)

2.2.1 New England in the Context of Mid-Latitude Land-Use History

The overall pathway of land-cover change in Massachusetts is not unidirectional in the face of ever increasing occupation from the Colonial era to present. Deforestation registered during the colonial frontier and subsequent agrarian phases of occupation. Forestation, however, marked the industrial phase, while forest fragmentation marks the advanced industrial-service sector phase and its suburban/peri-urban settlement patterns. This pattern is broadly representative of land cover changes throughout the northeastern and upper Midwest of the United States, although the dates of the transformations and the duration of landscape states vary. For example, over the last 30 years the upper peninsula of Michigan has witnessed an increase in forest cover from regrowth on abandoned farmland, but recently there has been an increase in fragmentation accompanied by a decrease in the size of ownership parcels driven by expansion of second home ownership and suburbanization (Drzyzga and Brown, 2002). This 30-year experience is similar to the 150-year one in New England. Western Europe also displays a shift towards more forest during the industrial era, and increased landscape fragmentation recently, although land management policies there may reduce the scale fragmentation found in the United States.

2.3 WATER WITHDRAWAL IN ARID/SEMI-ARID LANDS: OWENS VALLEY

The land history in the semi-arid Owens Valley, California, mirrors in many ways that observed in mid-latitude temperate regions world wide, in this case, driven by competing demands on its water sources (Putman and Smith, 1995) and illustrated in a time-line of major events in Figure 2. Though situated in a high desert of the Great Basin, the Owens Valley’s abundant and reliable water supply (from the surrounding mountains) encouraged establishment of agriculture beginning in the late 1800s, consisting of irrigated pasture and crop lands, including orchards. Water for agriculture was obtained by diversion of the
Owens River, and by the early 20th century, Owens Lake had begun to decrease in size and volume due to this diversion. Agricultural activity peaked in the 1920s, followed by large-scale abandonment due to a reallocation of the water resources, through inter-basin transfer, for the agricultural, domestic, and industrial demands of Los Angeles. Much of the abandoned agricultural land in the Valley was colonized by a mixture of perennial shrubs and annual grasses and plants. Water from Owens Valley, including the Mono Basin, makes up a significant fraction of the fresh water budget for Los Angeles, and all of the surface runoff has been exported from the valley since the 1920s. With the completion of a second aqueduct in 1968, the surface water export was supplemented by groundwater. With a diminished local supply of water, only a small fraction of the Owen Valley is cultivated today.

These transformations in water use and allocation have left a distinctive mark on the land cover of Owens Valley. The entire ecosystem downstream of the point where all surface water is diverted to fill the Los Angeles aqueduct has been transformed. The riparian and phreatophytic communities along the now dry Owens River have largely disappeared and the Owens Lake, once 280 km² in area, is now dry and constitutes the largest source of fine particulate aerosols (PM10) in the United States, posing significant health risk (Reheis and Kihl, 1995). The increased reliance on groundwater beginning in the 1960s caused many natural springs in the Valley to dry up, further reducing the amount of phreatophytic land cover (i.e., wetlands). Detailed studies (Elmore et al., 2003a; 2003b) of the resilience of...
the Owens Valley semi-arid ecosystems to the combined effects of a prolonged 6-year drought and the responses taken by resource managers show the following. (1) Phreatophytic communities are highly sensitive to depth to groundwater and show a threshold in response when water levels decrease below their rooting zone (3.3 m). Once this threshold is exceeded, the land is typically colonized by invasive shrubs and annuals, changing the ecosystem structure. (2) There is a legacy of land use. Abandoned agricultural land has lower species diversity and greater proportions of invasive shrubs and annuals, and this persists today nearly a century after abandonment.

The land-use and land-cover history of Owens Valley begins with an expansion of agricultural land use capitalizing on water resources. Agriculture contracted with the re-allocation of water resources for export from the region, outbid economically and politically by needs of Los Angeles. A period of relative stability in land cover and water abundance followed until additional demands were placed on the available water through groundwater extraction. The demands on water resources are now very close to the available supply, such that during periods of drought there is insufficient water for both natural and human needs. The net effect over the last hundred years has been the drying of Owens Lake, an expansion of invasive shrubs and annuals at the expense of native ecosystems, and a decline in wetlands. Periods of relative stability have been punctuated by short periods of water stress in which demand exceeds supply. This pulsed stress triggers important impacts (Elmore et al., 2003a).

2.3.1 Owens Valley in the Context of Semi-Arid Land-Use History
There are parallels between this environmental history and other arid and semi-arid regions in the United States. For example, the Great Plains saw an expansion of population and land under cultivation in the early 1920s, followed by a collapse precipitated by the dust bowl of the 1930s and a contraction in the amount of land under cultivation (Worster 1979). Economic changes and government policies allowed for several periods of expansion and contraction over the last 50 years (Brooks and Emel, 1999; Riebsame 1990). While the specific processes and drivers differ from region to region, the common threads are anthropogenic land transformations driven by water re-allocation or access (e.g., irrigation, diversion, export).

3 Land-Use Land-Cover Change Trajectories

Common LCLUC trajectories can be expected given an initial undisturbed state, three of which were detailed above. These trajectories involve four broad categories of land cover.

- **Undisturbed**: Landscapes dominated by “natural” cover types, where change is primarily by natural disturbance with little anthropogenic use (e.g. Amazonia in the 19th Century, New England in the 16th Century)(see note #1).

- **Frontier**: Landscapes experiencing transformations in “natural” cover, usually by extensive anthropogenic land uses (e.g., conversion to agriculture, forest re-growth through resource extraction) (e.g., Amazonia in the late 20th Century, New England in the 18th Century).
Agricultural/Managed: Landscapes in which management matches or supercedes nature in function, such as rangelands or cultivated lands sustained by intensive inputs. Land covers may be relatively stable, and changes in them are slow.

Urbanized/Industrialized: Landscape dominated by residential, commercial, and industrial land cover, and highly managed vegetation for services and recreation (e.g., parks, sports fields, and managed “natural areas”), but few resources of the land are utilized.

Most of the world’s lands can be categorized according to this broad framework, or some version of it, and significant portions have experienced one or more transitions from the undisturbed state. Where LCLUC histories are sufficiently long and well documented, it is possible to track a region’s transformation between these broad categories (Figure 3). A typical, full progression first involves a concerted movement of humans into the undisturbed landscape, motivated by push and pull factors, including natural resource extraction (forested systems) or agricultural colonization, or both. Where appropriate climates and soils exist, conversion to a managed landscape occurs, typically through explorations under extensive uses, followed by a contraction in the amount of land actively managed, due to poor economics and low returns, to that most economically viable. The abandoned land is usually re-colonized by natural cover, though with a species composition and ecosystem structure that is different than the undisturbed system. Intensification (greater inputs of labor, fertilizer, and other amendments) of the remaining actively managed land is a typical effect during this period. In those conditions favoring the emergence of an industrial-urban economy, non-agricultural land uses typically outbid agricultural uses, and a new period of land-cover fragmentation may be driven by urban expansion and suburbanization. This last phenomenon is perhaps more common in North America and Western Europe, but examples appear elsewhere, such as in the Pearl River delta of southern China.

This framework can be used to understand current conditions, past evolution, and future possibilities for land-cover change (Table 1). It is important, however, to clarify that not all areas have experienced or necessarily will experience the last two states noted and the time periods for any given period or transition is elastic. For example, logging in the boreal forest regions of Canada and Siberia, or in the mountainous regions of the Pacific Northwest of the United States are not activities meant to open up land for agriculture, but rather the logged lands are to be replanted or reforested for future harvest. Such regions may never become widely settled and/or urbanized, but will remain in an anthropogenically driven cycle of natural cover, deforestation, and regrowth. Likewise highly productive agricultural lands distant from densely populated regions and centers are unlikely to witness a transition to urbanization and suburbanization in the near future and may exist in a stable managed state for long periods. In much of the developing world, where rural populations have few options for food production besides extensive farming of marginal lands, abandonment and transition to more intensive agriculture is unlikely to occur without major changes in land tenure and economic conditions. Finally, all regions do not move unidirectionally through the four states. Southern Yucatán and much of Petén, Guatemala, for example, transitioned into the agricultural/managed state before A.D. 900, only to revert to tropical forest for a millennium before experiencing a frontier state today (e.g., Turner et al., 2001).
The transformation to a largely industrialized-urbanized state may be an endpoint in landscape evolution. There are no examples of an urbanized landscape of the magnitude having reverted to any of the previous states. Any large-scale de-urbanization would have to be accompanied by large reductions in human populations perhaps by relocation, war, famine, or economic collapse. While these may occur in the future, we have no examples of previously urbanized landscapes of the scale that exist today.

It is the transition between these generalized landscape conditions where the largest impacts of land use land cover change are manifested. The specific forces (drivers) of change that precipitate these transitions may vary by region and surely do in terms of their relative roles. For example, the transition from an undisturbed to a frontier landscape could be motivated at a national level by population pressures in a distant managed/urban landscape or the desire to secure sovereignty over remote land. Other drivers include policies to subsidize an extractive economy or to motivate individuals to develop subsistence or market agriculture. Specific impacts from the land transformations are documented in the context of case studies or cross cutting themes elsewhere in this book and include very evident biotic, biogeochemical and physical changes in the landscape (e.g., hydrology, nutrients, erosion, biodiversity, biomass carbon) as well as changes in the resilience of systems to interannual and interdecadal climate variability.

### 4 Quantifying Impacts

Terrestrial ecosystems provide many important goods and services on which human and other life depends, including regulation of climate, protection of watersheds, soil fertility, habitat to maintain diversity of plant and animal species, and cultural and aesthetic opportunities (Daily 1997; Ayensu et al., 1999; Daily et al., 1999). Primary among these goods and services is the provision of water, food and fiber. The vast majority of land use change is associated with conversion of undisturbed landscapes to cropland, either for local consumption or export to market (Geist and Lambin, 2002). Land-use change is essentially a trade-off between modifying terrestrial ecosystems for the positive benefit of providing food and fiber for human consumption and possible negative repercussions on other ecosystem services. These repercussions vary depending on the location and the state within the land-cover trajectory outlined in Figure 3. A focus of LCLUC research is to understand these impacts so that the trade-offs among ecosystem services can be quantified and assessed. The major impacts of LCLUC on ecosystem services are discussed below.

<table>
<thead>
<tr>
<th>Region</th>
<th>Frontier</th>
<th>Agricultural/Managed</th>
<th>Settled/Industrial</th>
<th>Post-industrial</th>
</tr>
</thead>
<tbody>
<tr>
<td>New England (US)</td>
<td>1650</td>
<td>1850</td>
<td>1940</td>
<td>2000</td>
</tr>
<tr>
<td>Western Europe</td>
<td>0</td>
<td>1100-1900</td>
<td>1850-1950</td>
<td>2000</td>
</tr>
<tr>
<td>Great Plains (US)</td>
<td>1860</td>
<td>1900-present</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rondonia (Brazil)</td>
<td>1960</td>
<td>2000</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Yucatán</td>
<td>0-200</td>
<td>900</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Siberia</td>
<td>2000</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
4.1 PROVISION OF FOOD AND FIBER

The predominant motive for land-use change is production of food and fiber. Although global food production is currently adequate to feed the world’s population (Lappe et al., 1998), many throughout the world either do not have adequate food and/or rely on low-yielding, unproductive land for their subsistence. Other regions have a surplus which is exported or used to grow animals for meat production. Two significant trends indicate that land-cover change for food production will continue into this century: increasing population in most of the developing world where people do not have the means to purchase food to satisfy their requirements; and rising incomes associated with increased
food consumption and diets richer in meat (Naylor 2000). The degree to which intensified agriculture with increasing yields can offset extensive agricultural expansion is a matter of debate (Tilman et al., 2001; Waggoner and Ausubel, 2001), but is ultimately a major factor for determining the amount of land-cover change.

4.2 ALTERATION OF BIOGEOCHEMICAL CYCLES

Land-cover change plays an important role in the carbon cycle, which in turn regulates the concentration of the greenhouse gas carbon dioxide in the atmosphere (c.f. Houghton et al., this volume). Expansion into “frontier” landscapes generally results in extensive clearing of natural vegetation; consequent burning and enhanced soil respiration results in release of carbon dioxide to the atmosphere. Model results estimate total carbon fluxes from human-induced land cover change of 188-192 Pg globally, with approximately one-third occurring prior to 1850 (DeFries et al., 1999; Houghton 1999). Since the beginning of the industrial revolution, land-use change has contributed approximately one-third of the total carbon released to the atmosphere from human activities, with 250 Pg of carbon released from combustion of fossil fuels (Fung et al., 1997). In past centuries, the frontier landscapes were generally in temperate grasslands and forests, but in the late 20th Century the last remaining frontiers suitable for cultivation absent climate change are the vast expanses of tropical forests in Latin America, central Africa, and Southeast Asia (Sanderson et al., 2002). The high biomass of these tropical frontier forests is of particular significance for carbon fluxes, with tropical deforestation comprising a substantial portion of the contemporary global carbon budget (Prentice et al., 2001). Although the precise contribution of tropical deforestation and regrowth is a major uncertainty (Achard et al., 2002; DeFries et al., 2002), the transition from undisturbed to frontier landscapes is a significant factor in the human alteration of the global carbon cycle.

With a transition from “frontier” to the “managed” state of the land-cover trajectory, higher-yield agriculture results from more intensive inputs of water and nutrients with profound impacts on nitrogen and phosphorus cycles. Use of synthetic nitrogen fertilizer, which supplements the natural processes that “fix” atmospheric nitrogen to biologically useful NH$_3$ and eventually to organic forms, has been one of the major factors responsible for increasing global food production and agricultural yields over the past several decades (Matson et al., 1997; Frink et al., 1999). Doubling of agricultural food production over the past 35 years was accompanied by a 7-fold increase in nitrogen fertilizer and a greater than 3-fold increase in phosphorus fertilization (Tilman et al., 2001). This anthropogenic alteration of the global nitrogen cycle has a number of repercussions, including the leakage of highly soluble nitrate (NO$_3$) from agricultural systems to cause eutrophication of surface waters, acidification of soil, groundwater pollution with nitrate, emissions of the greenhouse gas nitrous oxide to the atmosphere, and decrease of biodiversity as plants that favor a rich N supply displace other species. Release of phosphorus also results in eutrophication of freshwater streams and lakes. Regarding the carbon cycle, cropland abandonment in the “managed” state can sequester carbon from the atmosphere with regrowing forest (Caspersen et al., 2000).
4.3 ALTERED CLIMATE REGULATION THROUGH BIOPHYSICAL INTERACTIONS WITH THE ATMOSPHERE

Local, regional, and global climate are affected by land use and land cover through several types of interactions (c.f. Bonan et al., this volume). The structure and density of vegetation influence the amount of absorbed incoming short-wave radiation (albedo) and the turbulent exchanges of momentum, heat, and moisture (surface roughness). Through the process of photosynthesis, plants transpire water vapor through their stomates and affect moisture fluxes to the atmosphere and consequently the balance between latent and sensible heat. Changes in vegetative cover can consequently alter surface fluxes of energy and water and modify surface climate.

Several modeling studies illustrate the sensitivity of climate to changes in vegetation. At the global scale, a simulation with extreme cases of unvegetated and vegetated land surfaces generated a two-fold difference in land precipitation and 8K cooling in mean seasonal temperature with a vegetated relative to an unvegetated surface (Kleidon et al., 2000). In temperate and boreal regions, changes in vegetation may be responsible for a slight cooling owing to an increased albedo as brighter surfaces become exposed (Hansen et al., 1995; Bonan 1997; Bonan 1999; Bounoua et al., 2002). In the tropics, where forest clearing has predominantly occurred in the last few decades, the clearing likely leads to a warmer, drier climate (DeFries et al., 2002). Many model simulations of clearing the Amazon forest show increased temperatures and decreased precipitation (Nobre et al., 1991; Sud et al., 1996). Results of atmospheric general circulation models suggest that tropical deforestation may also influence climate through altered large-scale circulation patterns (Chase et al., 2000).

The feedbacks from land-cover change to climate through these biophysical mechanisms occur on spatial scales from local to regional, and possibly global through altered atmospheric circulation. The type of impact depends not only on the extent of the land-cover change but also where it occurs. During the “frontier” stage in temperate latitudes, the predominant effect was to cool surface temperature from an increase in albedo with land-cover clearing. In the current phase of frontier expansion in the tropics, the opposite is the case due to a large decrease in evapotranspiration associated with clearing of tropical forests (DeFries and Bounoua, in press).

4.4 WATERSHED PROTECTION AND SOIL EROSION

Changes in land cover alter the water yield and discharge for watersheds at all spatial scales from 10’s to 10,000’s of km$^2$ (Sahin and Hall, 1996). The canopy and root systems of vegetation affect a range of processes in the hydrologic cycle such as interception, percolation, surface retention, transpiration, and consequently surface and subsurface runoff and stream flow (Chang 2003). Rapid runoff, downstream flooding, soil erosion, and sedimentation are clear examples of local impacts of land-cover change. With transformation from undisturbed to extensive agricultural expansion in the frontier stage, examples of these local impacts include cropland expansion in eastern North America accompanying European colonization (DeFries 1986) and current clearing in the Amazon Basin (Williams and Melack, 1997). With a transition to more intensive production, these impacts would be lessened though nutrient exports would likely be enhanced (Mustard and...
Fisher, 2003). In the final urbanized stage, however, impervious surfaces will increase runoff, downstream flooding, and stream bank erosion.

4.5 FRAGMENTATION OF THE LANDSCAPE AND HABITAT LOSS FOR BIODIVERSITY

Habitat loss is the single greatest threat to biodiversity and is likely to be more significant for biodiversity loss than climate change in this century (Sala et al., 2000). Biodiversity is fundamental to ecosystem services by providing a genetic library as the basis for modern agriculture, medicine, and industry (Myers 1997). A growing literature is also establishing the importance of biodiversity for maintaining healthy, stable, and functional ecosystems (Chapin et al., 2000), in addition to the intrinsic ethical concerns about human dominance over nature.

As landscapes move through the trajectory from undisturbed and eventually to urbanized/industrialized, nature reserves and protected areas are critical for maintaining biodiversity, particularly in “hotspots” of endemic species (Myers et al., 2000). Reserves are generally successful in controlling land-cover change within their boundaries (Bruner et al., 2001), although they may be influenced by adjacent disturbed areas, particularly by atmospheric and hydrologic interactions. Even with the presence of nature reserves, rapid expansion of cropland in landscapes in the second stage of the trajectory can affect biodiversity, for example, by altering critical seasonal habitat for wildebeest in east Africa (Semeels and Lambin, 2001). Land cover change in the third or fourth stages of the trajectory can also affect biodiversity, for example the effects on bird populations from the construction of affluent rural homes in the Greater Yellowstone Ecosystem (Hansen and Rotella, 2002). As landscapes move along the trajectory described in Figure 3, resources, mobility, and interest in recreation increase, on one hand generating the demand for preserving landscapes but on the other hand placing heavy demands on the landscape for recreational use.

4.6 CULTURAL AND AESTHETIC OPPORTUNITIES

Land-cover change profoundly affects the aesthetic and cultural value associated with landscapes of all kinds. In the early stages, the cultural value largely derives from direct dependence on the ecosystem services (Gadgil and Guha, 1992). In the latter stages [states], society values and has resources to invest in recreational and aesthetic opportunities.

The above discussion illustrates that the nature of the impacts of land-cover change and the spatial and temporal scale over which they occur depend largely on the stage within the general trajectory described in this paper. As landscapes move through the trajectory from the “undisturbed” to the “frontier” category, the extensive clearing provides food and fiber mainly for local consumption. The clearing, however, has global and regional repercussions by releasing carbon previously stored in the vegetation to the atmosphere, altering climatic patterns, and reducing biodiversity through habitat loss. More locally, the clearing can generate soil erosion and increase runoff from reduced vegetation in the watershed. With agricultural intensification, the biogeochemical cycles associated with nitrogen and phosphorus are affected to a greater degree, and a decrease in cropland area can sequester carbon from the atmosphere and benefit biodiversity. In the final
urbanized/industrial state, the impacts are displaced in space as resources to support the population are obtained from afar, a spatial disjuncture that has proven difficult to incorporate into models.

5 Conclusions

Do LCLUC studies reveal broad commonalities in trajectories and impacts of land change? We conclude in the affirmative, and make the case for four general land cover/use conditions or states: Undisturbed, Frontier, Agricultural/Managed and Industrial/Urban. Many landscapes transition through these four states, though the timelines are elastic and there is no expectation that a given region is fated to experience all conditions. Furthermore the timeline is not unidirectional and through processes like abandonment land cover may revert from managed to undisturbed given enough time. The most profound impacts on land cover occur during transitions between conditions. This broad framework nevertheless masks many important details and its applicability to particular locations requires further investigation. This is particularly true with regards to the socio-economic drivers as the study of the linkages between land-use drivers, biological and physical impacts, and feedbacks to land-use decisions is yet in its infancy. Better understanding of these linkages, and the consequences for ecosystem services, will provide a basis for rational decisions about land-use change.

An ultimate goal of the LCLUC program is to affect policy and the framework presented here represents a beginning model for decision makers. For example, once the condition of a landscape is assessed, the pathways and attendant impacts can be linked to policy choices, recognizing the abundant uncertainties involved. Through the specific examples of the case studies and the cross-cutting themes that emerge from these studies, the land-change research community is honing the ability to articulate options and their outcomes.

Scale becomes an extremely important issue for quantifying impacts. Impacts that can be identified and characterized at a global or regional scale have had a great effect in framing questions and pointing to the magnitude of some problems (e.g., deforestation, land degradation, drought). Nevertheless, detailed characterization and quantification of impacts have generally relied on higher resolution observations typically at the scale of one ha or less. Impacts can be divided into those that affect the local environment (e.g., water quality) and those that extend far beyond the local environment (e.g., carbon, climate). This dichotomy of scale clearly hampers the development of an integrated understanding of LCLUC processes and impacts across space and time. In the NASA LCLUC program, many of the analyses have been at the spatial scale of Landsat Thematic Mapper for the central reason that this resolution is a good compromise between high frequency, low-spatial resolution global sensors and high-spatial resolution and large data volume but low temporal resolution sensors. While this TM-based perspective (space and time) has clearly led to important advances in identifying LCLUC pathways and impacts, there is a critical intermediate scale, the regional view, that needs to be addressed. The advent of new high spatial resolution sensors and more frequent observations coupled with expanding capacity to analyze data will likely lead to a better merging of local and global approaches in the future.
It is important to assess what impacts of LCLUC can be quantified. When considering the range of case studies, it seems clear that in regions with rapid and distinct changes in land cover (e.g., forest to cleared/agriculture, agriculture to urban), including rates, patterns, and trajectories, can be quantified by current approaches. Changes in the biophysical properties of the surface (e.g., live cover in semi-arid regions, woody vegetation encroachment) can also be quantified with some measure of success. Impacts of intensification (e.g., water quality) and changes in some land use, as well as land cover are possible though this has not been widely demonstrated. Some critical measures of landscape health will not be amenable to analysis with remotely sensed data. For these situations, and for incorporating socio-economic data, LCLUC analysis will have to rely on in-situ data and models parameterized by empirical relationships instead of direct parameterization.

6 References


CHAPTER 25

INTEGRATED LAND-CHANGE SCIENCE AND ITS RELEVANCE TO THE HUMAN SCIENCES

B. L. TURNER II, EMILIO MORAN, RONALD RINDFUSS

1Graduate School of Geography and George Perkins Marsh Institute, Clark University, Worcester, MA 01610
2Center for the Study of Institutions, Population and Environmental Change, Indiana University, Bloomington, IN 47405
3Department of Sociology, University of North Carolina, Chapel Hill, NC 27599

1 The Programmatic Origins of Integrated Land-Change Science

What is and ought to be humankind’s relationship with nature? This question has stood the test of time as an overarching intellectual and moral query confronting society and to which much research and pedagogy has been directed. The question can be traced to antiquity in western society (Glacken 1967), and has had no less profound thinkers in eastern societies. It has been recrafted in many forms following the Enlightenment, traced through such landmark concepts as noösphere and biosphere (Vernadsky 1945; Lapenis 2002), human modification of the earth (Thomas 1956; Marsch 1965), and ecosystem and biosphere function (Worster 1977; Lovelock 1988; Moran 2000; Golley 1992). These questions moved to the forefront of public concern in the 1960’s American environmental movement, inspired in no small part by Rachel Carson’s Silent Spring (1962), and led such initiatives as the International Biological Programme (IBP), which took advantage of the growing capabilities of computing to carry out large-scale ecosystem studies, including a “human adaptability” component examining the genetic, physiological, and behavioral adaptations that made it possible for human populations to thrive in environments considered to be extreme (Baker and Weiner, 1966; Odumi and Pigeon, 1970; Odum 1971; Baker and Little, 1976; Jamison et al., 1978).

UNESCO’s Man and the Biosphere Programme gave an even stronger role to the human dimensions of environmental concerns, especially as it has evolved today towards themes of sustainable development (www.unesco.org/mab).

Subsequent concern with global environmental change elevated questions to the structure and function of the biosphere, spurred in part by the incipient recognition of potential human-induced climate warming in the 1980s (e.g., Schneider 1989;

1 Global change science does not necessarily imply that all questions and analyses take place at the global or earth system scale. As Turner and colleagues (1990) noted, early in its development this science addressed both biogeochemical cycles operating in a globally fluid system and state changes, operating locally, that cumulatively reached a global magnitude. In either case, the critical causes and consequences are often highly localized and must be addressed accordingly. Subsequently, the global change and sustainability communities have amplified this last theme, seeking ways to insert “place-based” and other spatio-temporal scales of assessment onto the research agenda (e.g., NRC 1999a).
Houghton et al., 1990) and various assessments and stock takings demonstrating that the human impact on the biophysical systems of the earth had reached unprecedented conditions with profound implications for society worldwide (Turner et al., 1990; Steffen et al., 2003); humankind had entered the “anthropocene” (Crutzen and Stoermer, 2001) and a no analogue state (Steffen et al., 2002). This recognition not only stimulated the Intergovernmental Panel on Climate Change (IPCC) to examine the reality and causes of human-induced climate change, but also its societal impacts (Tegart et al., 1990). In 1986, the International Council for Science created the International Geosphere-Biosphere Programme (IGBP) to examine the systemic dynamics between the land, oceans, and atmosphere (Steffen et al., 2002). In its early development, the IGBP focused overwhelmingly on questions and issues of earth system science, with scant attention to the role of human behavior. Other organizations sought to create a human dimensions of global environmental program: internationally, the International Social Science Council (ISSC) which in 1990 created the forerunner of the Human Dimensions Programme (www.ihdp.uni-bonn.de), and in the US, the Social Science Research Council’s Committee for Research on Global Change and the National Research Council’s Committee on the Human Dimensions of Global Environmental Change which issued Global Environmental Change: Understanding the Human Dimensions in 1992 (Stern et al., 1992). It subsequently became clear to the IGBP, especially to its land components, that understanding the “human drivers” of land change was a critical but missing element of its science, and the IGBP and ISSC determined in 1991 to develop a joint effort on Land-Use/Cover Change (Turner et al., 1991). This decision had profound implications for the social sciences because it inserted basic research on the human causes and consequences of land change into the global change agenda (IGBP-IHDP 1999) paving the way for subsequent coupled human-environment studies of various kinds, including NASA’s Land Cover and Land Use Change (LCLUC) program, which taken together mark the emergence of integrated land-change science (Turner 2002).

Such integrative interests continue to enlarge as global change science matures and expands to issues beyond climate change, including questions of ecosystem services and health, biotic diversity, land degradation, and coupled human-environment consequences (e.g., Daily 2000; NRC 2001; Balmford et al., 2002). This last theme, captured under the label of sustainability science, garners increasing attention as the science, policy, and public communities turn to the “so what” issue (Raven 1997; 2 Various parallel and complementary programs internally and in the US also place land change high on their research agenda (e.g., NRC 2001). Examples include Millennium Ecosystem Assessment (www.millenniumassessment.org), PLEC (Population, Land Management and Environmental Change, www.unu.edu/env/plec/), National Science Foundation (US) program on Biocomplexity (www.nsf.gov), and the National Institute of Child Health and Human Development (NICHD) program on population and environment (http://grants1.nih.gov/grants/guide/rfa-files/RFA-HD-95-002.html). 3 This science is not unidirectional in orientation. In addition to human influences on biophysical systems, biophysical impacts on human systems are also considered. These multidirectional orientations are variously termed “reciprocal relationships” by social scientists and “feedbacks” by biophysical scientists. Language aside, the important point to remember is that an integrated land-change science needs to encompass the effects of land-use change on human behavior. While mounting evidence exists that land-use changes contribute significantly to global warming, the ultimate effects of global warming on humans will take some time to sort out.
Sustainability promises to engender research attention on coupled human-environment systems, promoting multi- and interdisciplinary programs and activities to the array of themes and issues dealing with the human-environment condition, its change and consequences (e.g., NRC 1999b; NRC 2001). As a concept, sustainability also extends beyond science per se, raising questions relating to values, policies, and competing interests, thus enlarging the sociological research side of sustainability.

1.1 THE COUPLED SYSTEM

Global environmental change science has improved our understanding of the dynamics of the biosphere and the consequences of human activity on the earth’s functions. Integrated environmental science or sustainability science is rapidly expanding the research agenda to questions of human impacts and policy (Lee 1993; Raskin et al., 1996; NRC 1999a; Buttimer 2001; Kates et al., 2001; Raven 2002). This expansion of the problem demands that the synergies between the human and biophysical worlds, or the coupled human-environment system, not only be considered but actually frame the research approach. This approach, in turn, fosters another kind of coupling, that of the heretofore largely discrete, if networked research domains (i.e., biophysical, social, and policy sciences) into explicit integrative and synthesis activities akin to “integrated assessments” (Smil 1993; Risby et al., 1996; Ehlers and Krafft, 2001; Rotmans and van Asselt, 2001).

Understanding the coupled system has long been recognized as a goal, if treated differently by different research communities. System-wide analysis has been hampered by various problems, however, ranging from the way in which research-pedagogy is organized to the paucity of coupled data to the inadequacies of computational and analytical techniques and tools. Improvements facilitating integrated research notwithstanding, an increasingly large community of researchers and decision makers realize that analysis of “coupled human-environment systems” cannot wait (Raven 2002), and considerable attention is being directed to the study of them (Moran 2000; NRC 1999a; 2000).

---

4 It is also important to remember that few social science data are truly global, and certainly not enough in terms of quality and concept-appropriate to sustain human-biophysical analyses and modeling at the global level. The availability of data on a country-by-country basis is related to factors that themselves are likely to affect global human-biophysical relationships. For example, more affluent countries tend to have more and better quality social science data than poorer countries. Countries with smaller populations (e.g., Norway) tend to have better data than those with larger populations. Countries whose political regimes that are outside mainstream international politics (e.g., North Korea) are likely to have poorer quality data, at least so far as it is available to the international research community.

Further, the data that do exist at the global level commonly have severe comparability problems. Much of this problem can be attributed to the paucity of global groups responsible and paying for the collection and compilation of the data. Instead the hundreds of units at the country and sub-national level who are responsible for collecting data do so with no or at best, modest coordination across data collecting units. Even when there is an elaborate and intensive effort to coordinate across countries, such as in the European Family and Fertility Survey project, critics complain about the lack of comparability (Festy and Prioux 2002).
“Coupling” connotes several intersections and linkages that are not always so obvious. For example, the LCLUC program and other human-environment coupling efforts have focused overwhelmingly on immediate or proximate linkages between cause and impact, fostering research on those systems in which these linkages are most obvious. While tropical deforestation is significant to global change, land degradation, loss in biodiversity and ecosystem well being are more tangible dimensions of this process. Tropical deforestation is commonly generated at the proximate level by semi-subsistence agriculturalists in which production and consumption decisions are intertwined within the same social unit (e.g., household) and in which the responses to community institutions (rules) and state policies can be readily observed. Yet, a dominant socioeconomic trend globally has been the decoupling of production and consumption decisions, both hierarchically and spatially. This process, commonly labeled “globalization” (e.g., Dicken 1992), is not only the subject of large interest in the social sciences, with roots extending back to Marx and Durkheim (see Axin and Barber, 2003), it commands that part of the economy worldwide that engender the most environmental changes. The coupled system addressed in land change and beyond, therefore, needs to account better for these more obscure and distal linkages, and it must develop techniques and models that can handle them (Kasperson et al., 1995; Schellnhuber et al., 1997; Wilbanks and Kates, 1999; Kasper and Kasper, 2001a).

1.2 INTEGRATED LAND-CHANGE SCIENCE

Integrated land-change science is pivotal to most, if not all, of the enlargement of global change and sustainability science. It seeks to understand the causes and consequence of land-change processes on the coupled system and subsystems through multiple ways of knowing (e.g., Buttimer 2001), but ultimately capable of understanding through modeling and other templates that speak to science and policy (Lee 1993; Schellnhuber and Wenzel, 1998). To achieve this goal requires a coupling of the biophysical, social, and GIS sciences into a common or integrative research framework.\(^5\)

The social sciences have long maintained various small-sized communities engaged in questions of the human-environment condition, ranging from resource economics and environmental policy to prehistory, and various subfields have claimed human-environment relationships are their subject of study (Turner 2002b). Despite this tradition, at least two facets of integrated land-change science have posed modest impediments to a larger social science entry into integrated land-change science. By definition, land-change science takes on problems that are enhanced by interdisciplinary, team-based research, whereas the social sciences commonly draw upon traditions promoting individual intellectual achievements; and but for a few exceptions, the social sciences have lagged behind the biophysical sciences in the use of satellite imagery and geographical information systems (GIS) for problem solving (see reviews in de Sherbinin et al., 2002; Liverman et al., 1998).\(^6\) These circumstances

\(^5\) GIScience refers to the use and development of spatial analysis through geographical information systems (GIS), including remotely sensed data (Goodchild 1992).

\(^6\) The NSF felt so strongly about this last issue that it supports the National Center for Geographic Information Analysis, and the Center for Spatially Explicit Social Science (U.C. Santa Barbara), to enhance the use of GIS, and to a lesser extent, remotely sensed data, among the social sciences.
appear to be changing, however. Large numbers of integrated research and teaching programs, both internationally within the US, are providing venues for social scientists and human-environment scientists to join biophysical and policy scientists in team-based, integrative studies, much of it land-change in kind (Turner 2002b). Moreover, junior social scientists are increasingly attracted to the use of GIS and remotely sensed data to assist in the problem solving, be it econometricians, demographers, political scientists, or agent-based modelers (Irwin and Geoghegan, 2002; Parker et al., 2002; Walsh and Crews-Meyers, 2002). For example, social scientists collaborated with biophysical and remote sensing/GIS scientists in generating the only official science documented presented by the United States to the Earth Summit in Johannesburg, South Africa, 2002, which was intimately linked to integrated land-change science (NRC 2002).

These developments notwithstanding, the usefulness of remote sensing data, especially that from satellite sensors, for the social sciences, especially in regard to their historic core concerns, remains problematic (e.g., Liverman et al., 1998; de Sherbinin et al., 2002; Rindfuss et al., 2003). This circumstance follows because the data in question reveal only some of the consequences of human decisions or socioeconomic structures, intentional or not, and their applicability for understanding the decision or structure in the first place has not been well demonstrated—not surprising given how recently these connections have been made between the data and tools provided by geospatial approaches and core questions of the social sciences.

In the remainder of this chapter we attempt to categorize the classical core concerns of the social sciences and consider their links to land-change studies and the likelihood that they might be addressed, in part, through remotely sensed data. This assessment is undertaken through an examination of process-pattern linkages that illustrate some of the possibilities of LCLUC and integrated land-change research to inform social science concerns. We conclude with comments on the future of the LCLUC program to serve more fully the human component of the coupled human-environment system of study.

2 Dominant Social Science Interests

Integrated land-change science stands outside or on the edges of the dominant or core concerns of the social sciences as they have emerged over the past century. The coupled human-environment system—a center piece of this science in which environment refers to the biophysical world—has not been addressed consistently throughout the social sciences for various reasons, including the negative impacts of late nineteenth and early twentieth century “environmental determinism” (see discussions in Moran 2000; Turner 2002b) and the attempts by the social sciences to free themselves from explanatory templates with origins in or strong connections to the natural sciences. This last effort is attested by the many challenges to post-positivism as an adequate explanatory framework for the social sciences (Guba 1990). To be sure, the coupled system has been addressed by the human-environment subfields (i.e., human ecology) as practiced in geography and anthropology (Moran 2000; Turner 2002), and various elements of the coupled system are examined in resource and environmental economics, especially among those subfields examining the human responses to natural hazards (White 1974 for work on natural hazards). The collective research of these
interests is substantial (e.g., Rayner and Malone, 1998), yet even a cursory examination of compendium of basic social science interests (e.g., *International Encyclopedia of the Social and Behavioral Sciences*) reveals this research comprises only a small portion of the overall attention. The overwhelming attention, of course, is cast to questions that resonate within and among such broad issues as agency-behavior, societal structures, and meaning.

Figure 1. Framework of Land-Change Dimensions Relevant to Social Sciences
(Note: overlapping quadrants represent various research efforts that fuse or cut across the binary labels employed here.)

Such core “human components” relevant to integrated land-change science can be framed in various ways. One way is captured in a three-dimensional matrix (Fig. 1) defined by the categories of cause-consequence (subject) on the X-axis, agency-structure (explanatory emphasis) on the Y-axis, and concept-application (orientation of contribution) on the Z-axis, the definitions of which are found in Table 1. Most social sciences enter environmental questions through concerns about the ways in which culture, economy, and political organization shape the perception and use of nature and the social consequences of interactions with nature. Land use and its human consequences are commonly explained by way of individual decision making and behavior (i.e., rational choice), political economic structures creating entitlements, opportunities, and constraints affecting decisions, or some combination of the two. A further division involves those practitioners focused on the origins of system properties or structures, and those concerned with functions of those structures. Regardless of which focus is taken, the underlying causes of land use—the factors affecting agency-behavior and structure—tended to mark the initial entry of the social science into land-change studies (e.g., Angelsen and Kaimowitz, 1999; Lambin et al., 2001; Ostrom et al., 2001; Geist and Lambin, 2002). With the enlargement of sustainability and integrated assessment themes, however, social science directed to the consequences of land change has enlarged to incorporate themes of natural hazards and vulnerability

---

7 Poststructuralist and postmodern perspectives largely deny a “metanarrative” that elevates the usefulness for agency-decision making or structures. Few practitioners of land-change science, however, adhere to these perspectives or attempt to incorporate them into modified “metanarratives” involving structures.
LAND-CHANGE SCIENCE AND THE HUMAN SCIENCES

The combination of subject and explanatory emphasis may be directed to concepts, themes, and theories of basic social science research or to real-world application, as in shaping land-use policies, although this distinction may be dissolving somewhat given the fusing of these orientations in sustainability science and integrated assessment.

Table 1: Description of Categories in Framework

<table>
<thead>
<tr>
<th>Category</th>
<th>Definition</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cause</td>
<td>Social, political and economic factors and processes influencing activities relevant to the use of land and its resources</td>
</tr>
<tr>
<td>Consequence</td>
<td>Social, political and economic outcomes of the land uses in question</td>
</tr>
<tr>
<td>Agency</td>
<td>Land-relevant behavior of and decisions made by manager of the land</td>
</tr>
<tr>
<td>Structure</td>
<td>Land-relevant rules and entitlements of opportunities and constraints on land manager</td>
</tr>
<tr>
<td>Concept-theory</td>
<td>Contribution to basic themes, theories, concepts of social science</td>
</tr>
<tr>
<td>Policy-application</td>
<td>Contribution to applied work and linked policy</td>
</tr>
</tbody>
</table>

Satellite data reveal much more about the consequences of land change than its causes, and speak less well to the questions of explanatory frameworks and concept-theory, than to various aspects of application. Recognizing these qualities, international agendas focused on land-change studies (e.g., IGBP-IHDP 1999) have reopened the pattern-process question: Are patterns of land change related to specific social processes and thus serve as indicators or measures of issues that strike to the core of social science interests (as noted above), either in regard to cause or consequence? Studies directed to this question under the label of “spatial geography” in the 1960s revealed that the same process could give rise to significant varieties of patterns, a not dissimilar outcome from the still earlier efforts of the culture-area approach in anthropology that found few correlations between biogeography and material culture (Kroeber 1939). With this caveat in mind, recent work hints that pattern-process relationships for certain bounded spatio-temporal land units exist (e.g., Brondizio et al., 2002; Evans and Moran, 2002; Moran et al., 2002), and the search is underway to uncover the implications of these relationships for theory and models (Irwin and Geoghegan, 2002; Parker et al., 2002; 2003). It is noteworthy that these findings point to one of the important contributions of the social sciences to an integrative land-use science: the significant variation in the combinations of biophysical and human processes operating in different locales and regions that give rise to land-use and land-cover change and the different response capacities of the coupled human-environment system in those locales and regions. For example, a recent meta-analysis of 152 case studies of tropical deforestation concluded that distinctive but different regional patterns exist in the causes and consequences of tropical deforestation (Geist and Lambin, 2002). While it is possible to extract some broad, general lessons about land-use transitions across the world and across land-use systems (Lambin et al., 2001), the local to regional variations are sufficiently large that models miss their mark if the differences are not considered. The social sciences have led the way in global change.
science in demonstrating the need to scale down from global to regional and local if the causes and consequences of environmental changes are to be understood (NRC 1999b).

3 Pattern-Process and Satellite Data

Remotely sensed data derived from satellite are a powerful source of information about the biophysical conditions of the biosphere, ranging from trace gas emissions to deforestation to urban expansion to the pathways that ENSO events take across Africa (Steffen et al., 2002). The abundance of information involved and the explosion of spatial analysis software (GIS) that permit the integration and analysis of this information with other data (Liverman et al., 1998) have rejuvenated the assessment of biophysical factors in human activities and outcomes (Rindfuss and Stern, 1998; Moran and Brondizio, 2001; Fox et al., 2002). The distinctive patterns in land change found at local and regional scales, and the power added to explanations and projections of land change by incorporating environmental variables into analysis have resulted in approaches that characterize human-environment relationships as interactive and that no matter the agency and structure involved in land management strategies, biophysical variables often play a profound role in shaping decisions and outcomes of land uses. This realization is not new and has been part of the centerpiece of long-standing research endeavors that cut across different explanatory perspectives. For the most part, however, they have not been accorded “core” status within the social sciences, and have been variously labeled the human-environment sciences (Kates 1987; Moran 2000; Turner 2002b). The significance of biophysical processes for understanding the coupled human-environment system notwithstanding, in the remainder we focus on those themes that have attracted the core attention of the social sciences, as noted above.

The social dynamics that affect land use and its change (e.g., land rents, zoning, globalization) and the societal consequences of land change (e.g., food shortages and security, land degradation and desertification, land tenure and concentration, and social unrest and justice) have long captured the attention of the social sciences, if not necessarily focused on the land-change question per se. What is more recent, and evident in the cases presented in this book and related work, is that some of these social dynamics and societal consequences can be inferred from the pattern-process connections revealed in satellite data (particularly when linked to appropriate social science data) and have been employed variously in research analysis and policy assessment (e.g., NRC 2002). Robust linkages require strong coupling to ground information, rigorous analytical assessment, and regular monitoring to confirm that the linkages in question remain operative or have changed. These caveats notwithstanding, the pattern-process understanding gained provides a powerful mechanism to address the spatial extent, magnitude, and rates at which certain social

---

8 Human-environment relationships are dynamic, changing as much for reasons that rest in shifts in social organization, political economy, technology, and wealth-poverty as in climate or other biophysical factors. Caution is raised among the social sciences in regard to those formulations of the relationship that focus only on the environmental factor or imply that stasis in this factor promotes stasis in the human consequences.
process are operating and societal consequences experienced. This knowledge, in turns, informs concepts and theories, models and projections, and integrated assessment and policy. Below we provide various examples, partitioned by cause and consequence, in which “core” social science interests are addressed by way of linked ground and satellite data and analysis.

3.1 SOCIAL DYNAMICS AFFECTING LAND CHANGE

[1] Why do sheep and game “farms” in parts of the Karoo of South Africa display such disparities in the quality of their land cover (biomass), given the relative uniformity of soils there? Archer (forthcoming) employs various statistical techniques to separate the climate and land-use signals in fine-tuned temporal AVHRR for lands dominated by white large-holders who have lost government subsidies in the post-apartheid government. This output is linked to farm survey data, revealing that lower quality vegetative cover (determined by NDVI measures) is strongly associated with those stockers following holistic range management strategies and that these strategies tend to be followed by those stockers of Afrikaner ancestry and those with debt load inhibiting experimentation with alternative strategies that are less reliant on cash flow from livestock itself. In this way, not only the regional impacts of stocking strategies are determined, but some of the “root” socioeconomic determinants land-use decisions are linked to these impacts.

[2] Various changes in the political economic structures of Mexico have significantly altered land uses and cover in the southern Yucatán, with potential impacts on the Calakmul Biosphere Reserve and associated programs. The Southern Yucatán Peninsular Region project (Turner et al., 2001; Turner, Geoghegan and Foster, 2003) links detailed imagery classification (Landsat TM) to reveal, among other factors, the patterning of lands under differing levels of use intensity. This information is linked to extensive and detailed household and other data to reveal the role of household land access on market involvement, and ultimately, on agricultural intensification and landscape consequences. For the most part, those communities with lower amounts of land per household are most strongly engaged in commercial cultivation, switching land-use strategies in which low-level capital inputs are attempting to follow a more permanent form of cultivation. The subtle landscape patterns detected from imagery analysis indicate where this economic orientation is taking place and its magnitude. Similar results have been demonstrated elsewhere in Yucatán (Sohn et al., 1999; Gurri et al., 2001; Gurri and Moran, 2002).

[3] Do households in Amazonia maintain constant rates of deforestation throughout their history? Studies of household lifecycles (demographic composition) in the Brazilian Amazon have been linked to satellite data revealing the role of gender and age structure of households in deforestation trajectories (McCracken et al., 1999; Moran et al., 2000; Brondizio et al., 2002; Evans and Moran, 2002; McCracken et al., 2002; Walker et al., 2002). Young households rapidly deforest their property (6%/annum) in their first five years of forest occupation as they seek to establish their farms and provide subsistence for the household. They steadily reduce the annual rates of deforestation with length of occupation, shifting land uses to more permanent crops
(e.g., cocoa, sugar cane) and pasture. As the households progress in their life cycle, deforestation briefly increases as farms are consolidated in preparation for production systems that will characterize their later years as an aging household (20-25 yrs in residence)—managing their fallows rather than undertaking new deforestation. This trajectory is affected by the biophysical conditions on the property. Households with fertile soils developed a more diverse portfolio of crops than those with infertile soils who appear to be forced into a mostly planted pasture strategy (Moran et al., 2002).

[4] Policy is invoked as the distal driver of land change, especially for tropical deforestation. Anderson (2000) demonstrates how the role of policy can be linked to Amazonian deforestation, determined by satellite data and ground surveys. In this region, policy directed to road building, credit, and fiscal incentives leads to initial deforestation—pushing the frontier—but subsequent, sustained deforestation is generated by local economic factors, including population growth on which policy has minimal impact. In short, the distal factor of state policy initiates deforestation but the control of policy vis-à-vis deforestation is soon lost to other, more local factors, such as community road building (also Walker et al., 1999).

[5] What role might “globalization” play in land change? This question has been addressed in regard to foreign direct investment and the spread of urban land uses in southern China. Proxies for this investment, coupled with other data, indicate a strong correlation with urban expansion as observed through satellite imagery, but this relationship is mediated by the local or proximate conditions of agricultural land productivity (Seto and Kaufmann, 2003). Given that some of the agricultural-to-urban land conversions involve some of the potentially most productive crop lands in China, identification of those conditions that give rise to conversion sheds light on policy options that would amplify or attenuate the loss of crop lands.

[6] Scalar relationships profoundly affect most assessments of land change. Modeling remote sensing and other data relevant to tropical deforestation in the southern Cameroon reveal the role of spatial inertia of change processes in which deforestation is amplified in lands adjacent to recent deforestation, giving rise to spatial spread effect and permitting assessments of the trajectories (magnitude and direction) of change (Mertens et al., 1997; Mertens and Lambin, 2000). As well, examinations in Nang Rong, northeast Thailand, have shown that relationships between population and environment depend on the scale of analysis (Walsh et al., 1999; 2001). Using remotely sensed data to measure land cover and plant biomass, GIS derived measures of elevation, slope-angle, and soil moisture potential, and social survey data for demographic data, it was found that at small-scales or fine resolutions, the relationships between social variables and land cover are strong. When coarser scales/resolutions were used, biophysical variables tend to maintain a strong relationship.

3.2 SOCIETAL CONSEQUENCES OF LAND CHANGE

[1] Southern Quintana Roo and Campeche, Mexico, constitute a hot spot of tropical deforestation, a land change process that threatens the Calakmul Biosphere Reserve (Turner et al., 2001). Owing to this threat and the aim to modernize production on communal lands throughout the country, shifts to neoliberal economic policies seek to
intensify extant cultivation and reduce the expansion of deforestation in the area. A state-sponsored, PROCAMPO provides direct payments to participating households based on existing cultivated lands maintained in crops and not permitted to return to forest. Household and community surveys, however, reveal that significant amounts of these payments are invested in clearing forest for pasture, in most cases absent livestock (Klepeis and Vance, 2003), and the total amount of land cleared for this purpose can be tracked by satellite data (Turner et al., 2001; Turner et al., 2003). The combined information suggests that this unintended, even perverse, consequence is prevalent among households and communities with “surplus” land, and appears to be undertaken as a means to lay claim to lands under conditions of tenure uncertainty and in hopes that state-led livestock programs will follow (Klepeis and Vance, 2003).

[2] May land preservation policies have adverse consequences for people and the land? Yes, according to work undertaken in the “woodland” landscapes of Rajasthan, India. Linking Landsat data with discourse assessment, Robbins (2001) shows that the woodlands are transforming to hybrid or “quasiforests” complete with exogenous species that have proven difficult to control and which have significant production consequences for the local occupants-land users. The hybrid woodlands follow from state planners holding the view that reduced herding and other activities would reduce woodland degradation, when in fact the qualities of the woodlands that the state sought to preserve required land use. Absent an understanding of the coupled system dynamics but holding the capacity to regulate land use (Robbins 1998), state policy has apparently had negative consequences for the woodlands and the people.

[3] As part of its international responsibilities to preserve endangered biota, China has established a Panda Reserve with the explicit aim of preserving the critical habitats that support this endangered species. Using Landsat data, Liu and his colleagues (1999) have demonstrated that the type of settlement pattern adjacent to or within the preserve has significant consequences for these habitats and, hence, the social aim of preservation. Surprising, even to some experts, was the finding that areas with the less dense but spatially dispersed settlements has a greater negative impact on panda habitat than areas of dense, concentrated settlement. This relationship appears to be related to the distances that local occupants are willing to travel to their fields and to collect wood fuel. This distance does not vary by settlement pattern, such that a dispersed pattern affects more of the critical panda habitat, while the dense pattern reduces the total habitat affected, even if its effect is more profound for the use area. The use of remote sensing data permitted an important, broad generalization that other methods would have been inadequate to grasp as quickly and as convincingly.

[4] Can land-change agendas affect the well being of locale people? A strong case can be made that policies aimed at regulating landscape burning in Mali are linked in this way. Despite the antiquity of human-controlled landscape burning there and elsewhere in western Africa, desertification and related land degradation themes have resulted in policies attempting to reduce the activity. Laris (2002) links interviews and study of those actually setting fires with remote sensing data to examine the human-environment rationale for the activity and policy impacts in Mali. Intentionally set fires tend to be small scale and follow a seasonal rhythm that creates a landscape mosaic of biomass that is less likely to trigger a large-scale burn that threatens homes and fields alike.
State policy is seen as reversing this circumstance, endangering the well being of the occupants.

[5] Various examples from the health fields illustrate human consequences linked to land change and other environmental considerations. Seto and colleagues (2002), for example, join biophysical and remote sensing data to project the change in the magnitude and location of schistosomiasis in southern China that will follow from the Three Gorges dam on the Yangtze River. Linthicum and colleagues (1999) advance such assessments by linking historical records of precipitation and Rift Valley Fever in Kenya, linking outbreaks of the fever to abnormally high rainfall. They then link ocean warming with such rainfall fluxes and, with remote sensing techniques, to map its spatial dimensions across the country. These same techniques permit a five-month advanced forecast of the epidemic. Similarly, Landsat Thematic Mapper data have been used to identify human risk to hantavirus pulmonary syndrome by estimating the location and expansion of sites favoring the deer mouse in 1998, following an ENSO event (Glass et al., 2002). These cases, illustrate how satellite data are combined with other information to create forecasts and projections that guide policy and planners in regard to potential health hazards.

4 Integrative Land-Use Science and Its Future with Remote Sensing

Integrated land-change science seeks to forge a union among biophysical, socioeconomic, and remote sensing information and understanding to address the coupled human-environment system. Here we have focused on the socioeconomic-societal and remote sensing coupling as it contributes to the long-held interests of the social sciences. We have attempted to illustrate that these linkages help to reveal various characteristics of both the causes and consequences of land change as they are understood through both agency and structure to address conceptual themes and application.

A frequently stated claim and concern speaks to the potential of remote sensing techniques generating and thus replacing traditional social science data, such as found in censuses, surveys, and face-to-face encounters with the people in question. Unless there are some startling breakthroughs in remote sensing technology, satellite data (and aerial photography) will never replace the data in question. Remotely sensed data can detect the physical characteristics of the landscape and the objects that comprise that landscape as well as various attributes of the objects (e.g., their dimensions and physical properties). It is obviously unlikely that remotely sensed data alone would ever be able to determine the number of people occupying that landscape, including their age, sex, race, education, occupation or other standard demographic variables, nor will it provide information about their attitudes, decision-making rationales, and the guiding political economic structures and policies for the observed land unit. The data needs of integrated land-change science will never be fully met by remotely sensed data.

Even though remotely sensed data will never provide the complete data needs of the land-use research community, the time is ripe to query the extent to which remotely sensed data can provide some variables that might be considered “social.” Examples include indexing the land context around human settlements and using night-
lights as a proxy of the measure of human occupational density. Judith Lessler once observed that in the early history of brick-making technology, bricks were made in the shape of stones – which is about where we are in using remotely sensed data to measure social science concepts. Just as brick technology advanced, however, we anticipate more creative uses of remotely sensed data to measure social science variables, but we suspect that these refinements will address the “physical” dimensions of those variables. While important, these dimensions and variables will be insufficient to address the coupled system of integrated land-change science. It is worth noting that, to date, a very small portion of global change research funds has been devoted to the collection and analysis of social science data linked directly to remotely sensed data. One result is that the proportion of projects in LCLUC and similar programs with a distinct and strong social science component remains regrettably small. Greater advances might be expected if investments commensurate with the importance of the human dimensions of global change were made.

The data linkages notwithstanding, a truly integrative land-change science requires improvement in the way students are trained so that they are given the experience and excitement of engaging integrative research in the lab and the field. Those who have led in this domain need to ensure that the benefits of this integrative science fertilizes the social sciences and leads them more fully into critical local to global concerns captured in integrated land-change science. The burden of our current community is to demonstrate that the coupled human-environment system is of fundamental importance to understanding the human condition on this planet. The burden of the social sciences at large is to realize that beyond their traditional cores, the broader science communities have elevated the significance of understanding the coupled human-environment system in integrative ways, reinvigorating the “great” question with which we began this chapter and portending to change the partitioning of knowledge, otherwise known as the academy (Kates et al., 2001; Turner 2002b).

5 References


CHAPTER 26

RESEARCH DIRECTIONS IN LAND-COVER AND LAND-USE CHANGE

ANTHONY C. JANETOS


1 Introduction

One of the conclusions from the previous chapters in this volume is that enormous progress has been made over the past decade in land-cover and land-use change research. The community has made great strides in implementing the original research agendas that have been described in international, national, and agency plans (Liverman et al., 1993; Turner et al., 1993; Turner et al., 1995; Janetos et al., 1996; IGBP 1999; Lambin et al., 1999).

But one could also conclude that the original motivations for the rapid development of research into land-cover and land-use change have yet to be fully realized. The original vision was to understand the end-to-end sequence of land cover and land use changes, integrating changes that are driven by natural variability with those driven by human decisions, measuring the actual changes on the landscape, and evaluating both the ecological and socioeconomic consequences for humans (Janetos et al., 1996). Moreover, the international and national programs have emphasized the need to develop models of the various processes involved, with the ultimate goal of developing integrated models that can simulate the important processes and consequences for particular landscapes or societies (Liverman et al., 1993; Turner et al., 1993; Turner et al., 1995; Janetos et al., 1996; IGBP 1999; Lambin et al., 1999).

The goal of this chapter is to present the major research challenges that have not yet been met, but that still will be required to achieve the ultimate goals. The chapter will address both theoretical and empirical issues, and will further address some of the methodological needs that future studies will have.

2 The Need for More and Better Data

One of the original challenges in the agenda for land-cover and land-use change research was the availability of data that had been collected in a way that allowed spatially explicit analysis to be done, or alternatively that could be used to calibrate or test the performance of integrated models. The availability of data remains a challenge. In the discussion that follows, we consider the availability of land-cover and land-use data themselves, and then consider the availability of socioeconomic and demographic data.
2.1 LAND-COVER AND LAND-USE CHANGE DATA

One of the primary goals for the Land Cover and Land Use Change programs is to develop the capability to perform repeated inventories of global land cover and to identify areas and quantify rates of land-cover change. The past decade has seen a rapid increase in collecting, analyzing and distributing quantitative information on land cover. In particular, the implementation of satellite-based measurement systems has been important in this respect. However, while there has been a large amount of data collected, there still remain critical issues of validation, data availability, and continuity of important data streams.

The past five years have seen a dramatic expansion of new satellite-based measurement systems, from both the European Union and from the US. With only a couple of exceptions, however, new surface measurements have been collected at scales of 1km ground spatial resolution. Global data sets of land cover have been produced using such data, combined with many other data sets, beginning with AVHRR, and now continuing with MODIS and SPOT-VEGETATION (Eidenshenk and Faunden, 1994; Loveland et al., 2000; Freidl et al., 2002; Stibig et al., 2003). However, the scientific community has known for some time that data at this spatial resolution are of limited use for quantifying such land cover changes as deforestation in the humid tropics. It remains to be seen whether there are intrinsic limits in the detectability of change using such data, even though the characterization and calibration of the sensors themselves has improved dramatically since AVHRR (Zhan et al., 2000).

There is new experience with very high spatial resolution data (c. 1-3m) from the private sector, and it is already clear that such information can be quite valuable for understanding very local-scale changes. But the amount of data needed for regional studies is prohibitively large for most scientific users, and the expense of acquiring the data also is large for most scientific budgets. Additionally, there are as yet no systematically acquired data sets of very high spatial resolution with which to build time series information for locations of particular scientific interest.

The chapters in this volume demonstrate clearly that Landsat TM and ETM+ data (or the equivalent SPOT data) continue to have tremendous scientific utility. Not only are the data themselves systematically gathered in such a way that long time-series can be constructed for most places on the Earth, but also the methodological tools for analysis are easily available (Goward et al., 2001). The Landsat global acquisition plan has resulted in expanded global coverage and multiple acquisitions each year (Goward et al., 2001). The spatial resolution of the Landsat data also correspond reasonably well with many socioeconomic data sets, at least those in which locational information has also been recorded, so that combined analyses can be performed. The total amount of Landsat data needed for large regional studies, while very large compared to most socioeconomic data, is still well within the technological capabilities of many research groups. The costs to users of Landsat data have declined in recent years, primarily when the program returned to the public sector, and for a small number of scenes, are now affordable within the budgets of most research groups in industrialized countries. For regional studies that require hundreds of Landsat scenes or multi-date analysis, data costs remain prohibitive to the research community. Recognition of this obstacle in part
prompted NASA to purchase and make available orthorectified global Landsat coverage for 1990 and 2000 to the research and development communities. NASA has also recognized the need for emphasis on improved data access and to this end has supported data distribution initiatives through the Global Land Cover Facility and the Tropical Forest Information Center.

However, while the utility of Landsat data is amply demonstrated within these chapters, there remain unfulfilled needs. One of the main features of the Landsat data record is its continuity over decades (since 1972). As of this writing, a new contract for a follow-on mission to Landsat 7, launched in 1999, has yet to be awarded, and technical difficulties are plaguing the current instrument. The earliest launch of a follow-on mission would be scheduled for 2007, two years past the design life of the ETM+ instrument on Landsat 7. Indeed the recent ETM+ instrument malfunction highlights the need for operational status of the high resolution instruments and the fragility of our terrestrial observing systems with serious consequences for the kind of interdisciplinary research highlighted in this volume.

This situation creates at least two strategic research challenges. One is to secure high resolution observations as part of the operational satellite systems and in the interim to investigate whether other sources of land-cover information can be used successfully in interdisciplinary studies (e.g. data from ASTER). The second is to ensure that all the current data can be assembled into time series of land-cover changes for important regions, if not globally, as quickly as possible. The utility of the latter will be several-fold. Such data, especially if they are nearly global in extent, would provide an enormously valuable validation data set for coarser-resolution remote sensing imagery that will continue to be collected much more frequently than will Landsat or equivalent follow-on missions. Second, when a follow-on mission is ultimately flown, especially if there has been a gap in the continuity of the data record, such global data sets will provide the only Landsat-scale reference data against which the new data can measure change.

At a more operational research level, the studies in this volume also demonstrate that methods of analysis for remote sensing data must continue to be improved. In regional studies, more progress must be made on automating the classification of the remote sensing data into different land-cover types and their changes. The validity of the conclusions from individual case studies of land-cover and land-use change obviously depend to an enormous extent on the accuracy of the underlying land-cover characterization. Such validity can generally only be provided by careful comparison with ground-truth data, along with local expertise of the analysts. Standard procedures for collecting validation data and reporting accuracy results is needed. A new initiative by the Committee of Earth Observations (CEOS) to develop international protocols is starting to address these issues. Continued effort into both increased automation of land-cover change detection and classification itself will enable interdisciplinary analyses to be conducted more efficiently, and with more confidence in the results.

The challenges of correlating land-cover data with land-use data raise additional concerns for research that seeks to achieve an interdisciplinary synthesis. Even in many of the studies in this volume, researchers are seeking to correlate spatially explicit data on land cover with what are essentially tabular data on land use, or at best, land-use information that is collected according to political or administrative
subdivisions rather than with explicit spatial information. The resulting mismatches of data can be potentially confounding in further analysis.

2.2 SOCIOECONOMIC DATA

The collection and provision of socioeconomic data in a form that can be matched with spatially explicit land-cover data were two of the challenges originally identified by IGBP/IHDP and more recently by the NAS. This volume demonstrates that this is still a challenge. Data that are collected specifically within the context of a particular study are much more likely to match the spatial scale of the land-cover information. However, a great deal of socioeconomic data comes from government institutions, which routinely collect them for a wide variety of reasons. Such information may match the spatial scale of available land-cover information, but generally such a match will be fortuitous. Analytical methods that can deal with this problem are clearly required. The availability of socioeconomic data from individual governments varies considerably. The raw socioeconomic data are rarely available, as there are issues of privacy and statistics are often summarized at the sub-national level. Spatializing these summarized data for use in spatially explicit models is often problematic.

There is a third class of data and information that is required, which falls into somewhat of a gray area between physical/biological land-cover data and socioeconomic data. This is information on particular features of the physical environment that are also critical for socio-economic analysis, but which are difficult to discern from imagery. For example, it may be quite difficult to distinguish one crop type from another on the basis of remotely sensed imagery alone, especially in areas where seasonal coverage is sparse, but knowing what local residents have planted and what the final yields are may indeed prove critical for understanding local land-cover and land-use change. The existence of road networks, particularly in rural locations, which are critical for access to both resources and markets, but may be difficult to discern on imagery, is another example. Both better data and better methods of analysis will be required to ensure that such information is available to scientists interested in these issues.

3 Better and Sharper Hypotheses of Causation

The studies in this volume are quite variable with respect to their success at achieving the interdisciplinary synthesis that has guided the NASA program and the scientific community. This is not surprising. Interdisciplinary synthesis is extraordinarily difficult, since it generally requires investigators to have in-depth knowledge of other disciplines. Such knowledge often takes years of collaboration before an interdisciplinary study can evolve.

In a newly developing area of science such as this, the documentation of change in land cover and land use, and its association with the human dimensions of cause and consequences is important (Turner et al., 2002). But it is also equally important to seek hypotheses about the observed changes that yield additional insights into the root causes of the changes, and that can lead to observations that can truly distinguish among alternative hypotheses. For example, it has been extremely
important in studies of deforestation in the Amazon to recognize the early influence of government policies that created financial incentives to deforest. But these policies have been repealed for some time, and deforestation rates continue to vary from year to year, and have reached annual rates that are every bit as high as previously. There are extensive multivariate analyses of the influences of population increases \textit{per se} and economic activity on deforestation rates. But the possible causal pathways of the various factors are extremely complex, including as they do socioeconomic factors within the regions of interest, the influence of external socioeconomic factors, such as the need for foreign capital from export crops, access to markets, and even the influence of interannual climate variability (Lambin et al., 2001; Geist and Lambin, 2002). We have yet to construct alternative hypotheses that are successful in a diagnostic sense; i.e. that are successful in their ability to distinguish one set or syndrome of causes from another and why.

Deforestation is but one example. Many of the studies in this volume address parts of the world in which the influences of local, national, and international socioeconomic factors and policies are extremely complex and difficult to distinguish on their own. In addition, the ways in which these factors interact with the biological and physical resources on the landscape is also complex, and often poorly explored. The early evolution of these interdisciplinary studies thus has necessarily focused on documenting what the changes have actually been, and what the association between potential causes and consequences may be. The next stage of investigation should be a much clearer focus on why the observed changes have occurred, and not others; or alternatively, why some combination of potential causes has resulted in particular consequences on the landscape or for people. Most of the studies of the processes of change are undertaken at the local level, for example through village or community surveys. The challenge is to be able to scale the processes up to the regional level, providing a more general and useful assessment and develop process-based predictions.

4 Better Models of Decision Making and Consequences

If the current generation of studies can fairly be described as being primarily oriented towards the description of land-cover and land-use change, with some initial exploration of causal hypotheses, then an obvious next step for research is the development of quantitative modeling that can be used either to understand observed changes or eventually to predict them. The scientific community’s current ability to construct and validate such models is limited in several ways. One is simply the availability of data either to construct or validate models. Because this chapter has already considered data availability at some length, it will not be discussed further.

Another limitation, however, is creating an adequate conceptual framework for modeling the actual decision-making process that people employ when making land-use decisions over time. The simplest framework is essentially economic; i.e. it assumes that any time, a decision-maker will maximize the present value of his/her resources. Value is generally defined in economic terms. But we know that economic value, while important, is not the only criterion that is important for decision-making. In many situations, especially in the developing world, access to markets is difficult, so other types of value may take precedence. In addition, other types of services that are
provided by ecosystems, e.g. the maintenance and regulation of water flows and quality, production of soil fertility, etc. are also important and are often recognized, at least implicitly, by decision-makers (Alcamo et al., 2003). There is also a large and growing literature on other possible criteria that may be used in making what are essentially cost-benefit decisions. These may include minimization of losses rather than maximization of net gains, increasing the predictability of gains, even if they are sub-optimal, and so on. And the influence of social and cultural factors is clearly important in the real world, but often difficult to capture in models.

Many studies would benefit from a fuller appreciation of who the decision-makers in land-cover and land-use change actually are. Having done that, a calculus and accounting of their costs and benefits for particular changes would be more illuminating. Each of the studies in the chapters in this volume, for instance, has within it an interplay among local, regional, and often national and international actors, each of whom has different goals, value systems, and rewards that they may gain from any particular land-cover and land-use change. Accounting for the maze of interactions among the various actors in any situation presents both empirical and conceptual difficulties, but may in fact be necessary for constructing reasonable models of both cause and consequences of land-cover and land-use change.

For the accounting and subsequent models to be complete, they will need to include those factors that are important in decision-making, whether economic, physical, biological, or cultural. They will also need to include a far more integrated description of the consequences of land-cover and land-use change, from economic, biophysical, and cultural perspectives, as well as describe how those consequences are distributed among different actors and over time. This is a tall order indeed, and suggests that the scientific community will need to take a stepwise approach to constructing and testing such models.

Providing the scientific underpinning to improved land-use decision-making and policy is one of the more challenging but essential goals of the Land Cover and Land Use Change research agenda. Increasingly the global change community is being asked to show the societal return for the investment being made in research. An explicit initiative is needed to develop this policy relevant dimension of the program, requiring rather different skills than the research itself. Achieving this goal will involve development of an on-going dialogue with resource managers and decision makers to better understand the management or policy information needs. Effective interfaces will need to be developed between scientists and resource managers creating a two-way flow of information on current scientific understanding and information needs.

5 Transition to Operational Monitoring

Because land-cover and land-use change is a continuing phenomenon, one of the largest challenges is how to build an ongoing ability to monitor its physical manifestations and couple them with ongoing measurements of their socioeconomic causes and consequences. Many, but not all of the economic data necessary for this are routinely collected by governments because they provide indicators of economic activity and productivity. But collecting the land-cover data themselves on an operational basis is an ongoing challenge. There are several international efforts in
place around the world, coordinated through such activities as GOFC/GOLD and CEOS, that may provide models of how such operational capacity can be built and maintained (Townshend et al., this volume). But ultimately, the transition to operational status of any of these monitoring efforts requires the commitment of stable institutions that can provide resources and the human capacity for collecting and distributing information to interested parties. In the US the continued role of NASA in prototyping the systems for providing space-based land-cover inventories that can be transitioned to the operational domain is essential. However, a commitment is now needed from the operational agencies to provide the long term monitoring of land-cover and land-cover change.

A further lesson on the transition to operational monitoring on land-cover and land-use changes is that land use itself needs to become a more carefully monitored activity. Unlike the energy system, where each transaction is also a market transaction and can therefore be tracked independently by financial means, each change in land cover or land use is not necessarily captured in markets. Some changes are driven by purely natural factors; others are driven by human decisions either within markets or outside markets. But without a full accounting over time of both land cover and the uses to which the land is being put, it will be very difficult to succeed in creating a complete description of causation and consequence, much less construct modeling capacity.

Because of the interdisciplinary nature of the issue, single research and/or operational agencies or institutions are likely to find the transition to operations to be a very difficult challenge. The space agencies by themselves, for example, do not have the expertise to know which socioeconomic data ought to be collected, on which scales, and how those data might best be archived, distributed, and used. Economic agencies and institutions are quite unlikely to have the expertise to make the same types of judgments about physical measurements of land-cover change itself. This suggests that new types of partnerships will be important for making a transition to operational status for this research area. The identification of Land Use and Land Cover Change as a new topic for global change research by the U.S. Government may provide some new interagency partnerships on research that will lead to global land-cover monitoring achieving operational status (CCSP 2003).

6 Conclusion

The achievements of the land-cover and land-use change research community have been significant, but the remaining challenges are considerable. Interdisciplinary teams of researchers capable of addressing integrative research questions will be needed. A new generation of scientists will need to be trained with an understanding of both physical and social science and our current discipline oriented university curricula will need to be adjusted accordingly. Development of improved observations and data sets, sharper hypotheses and better integrated models will take a concerted community effort. Programs such as the NASA Land Cover and Land Use Change program, which has focused largely on the observations, now need to be augmented by initiatives from other agencies and organizations, strengthening the socio-economic analyses and the integrative and predictive modeling aspects of the research. For the remote sensing
community, establishing the operational flow of data and information and land cover and change, developing advanced information products and strengthening decision support systems with reliable observations are priority areas for research.

The pressing societal questions associated with increasing pressure on the land from economic and demographic changes and an increasing variability in climate make the study of land-cover and land-use change a priority. Developing a scientific understanding that can be used to inform policy and decision-making, leading to improved use and management of the land is imperative if we are to develop a sustainable future for life on this planet.

7 References


