Chapter 2

Analytical Approaches for Assessing Ecosystem Condition and Human Well-being

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Main Messages

Many tools are available to assess ecosystem condition and support policy decisions that involve trade-offs among ecosystem services. Clearing forested land, for example, affects multiple ecosystem services (such as food production, biodiversity, carbon sequestration, and watershed protection), each of which affects human well-being (such as increased income from crops, reduced tourism value of biodiversity, and damage from downstream flooding). Assessing these trade-offs in the decision-making process requires scientifically based analysis to quantify the responses to different management alternatives. Scientific advances over the past few decades, particularly in computer modeling, remote sensing, and environmental economics, make it possible to assess these linkages.

The availability and accuracy of data sources and methods for this assessment are unevenly distributed for different ecosystem services and geographic regions. Data on provisioning services, such as crop yield and timber production, are usually available. On the other hand, data on regulating, supporting, and cultural services such as nutrient cycling, climate regulation, or aesthetic value are seldom available, making it necessary to use indicators, model results, or extrapolations from case studies as proxies. Systematic data collection for carefully selected indicators reflecting trends in ecosystem condition and their services would provide an improved basis for future assessments. Methods for quantifying ecosystem responses are also uneven. Methods to estimate crop yield responses to fertilizer application, for example, are well developed. But methods to quantify relationships between ecosystem services and human well-being, such as the effects of deteriorating biodiversity on human disease, are at an earlier stage of development.

Ecosystems respond to management changes on a range of time and space scales, and careful definition of the scales included in analyses is critical. Soil nutrient depletion, for example, occurs over decades and would not be captured in an analysis based on a shorter time period. Some of the impact of deforestation is felt in reduced water quality far downstream; an analysis that only considers the forest area itself would miss this impact. Ideally, analysis at varying scales would be carried out to assess trade-offs properly. In particular, it is essential to consider nonlinear responses of ecosystems to perturbations in analysis of trade-offs, such as loss of resilience to climate variability below a threshold number of plant species.

Ecosystem condition is only one of many factors that affect human well-being, making it challenging to assess linkages between them. Health outcomes, for example, are the combined result of ecosystem condition, access to health care, economic status, and myriad other factors. Interpretations of trends in indicators of well-being must appropriately account for the full range of factors involved. The impacts of ecosystem change on well-being are often subtle, which is not to say unimportant; impacts need not be drastic to be significant. A small increase in food prices resulting from lower yields will affect many people, even if none starve as a result. Tracing these impacts is often difficult, particularly in aggregate analyses where the signal of the effect of ecosystem change is often hidden by multiple confounding factors. Analyses linking well-being and ecosystem condition are most easily carried out at a local scale, where the linkages can be most clearly identified.

Ultimately, decisions about trade-offs in ecosystem services require balancing societal objectives, including utilitarian and non-utilitarian objectives, short- and long-term objectives, and local- and global-scale objectives. The analytical approach for this report aims to quantify, to the degree possible, the most important trade-offs within different ecosystems and among ecosystem services as input to weigh societal objectives based on comprehensive analysis of the full suite of ecosystem services.

2.1 Introduction

This report systematically assesses the current state of and recent trends in the world's ecosystems and their services and the significance of these changes for human livelihoods, health, and well-being. The individual chapters draw on a wide variety of data sources and analytical methods from both the natural and social sciences. This chapter provides an overview of many of these data and methods, their basis in the scientific literature, and the limitations and possibilities for application to the assessment of ecosystem condition, trends, and implications for human well-being. (See Figure 2.1.)

The Millennium Ecosystem Assessment's approach is premised on the notion that management decisions generally involve trade-offs among ecosystem services and that quantitative and scientifically based assessment of the trade-offs is a necessary ingredient for sound decision-making. For example, decisions to clear land for agriculture involve trade-offs between food production and protection of biological resources; decisions to extract timber involve trade-offs between income from timber sales and watershed protection; and decisions to designate marine protected areas involve trade-offs between preserving fish stocks and the availability of fish or jobs for local populations. Accounting for these trade-offs involves quantifying the effects of the management decision on ecosystem services and human-well being in comparable units over varying spatial and temporal scales.

The next section of this chapter discusses data and methods for assessing conditions and trends in ecosystems and their services. Individual chapters of this report apply these methods to identify the implications of changes in ecosystem condition (such as forest conversion to cropland) for ecosystem services (such as flood protection). Rigorous analyses of these linkages are a key prerequisite to quantifying the effects on human well-being (such as damage from downstream flooding).

The third section discusses data and methods for quantifying the effects of changes in ecosystem services on human well-being, including human health, economic costs and benefits, and poverty and other measures of well-being, and on the intrinsic value of ecosystems. These methods provide a framework for assessing management decisions or policies that alter ecosystems, based on comprehensive information about the repercussions for human well-being from intentional or unintentional alteration of ecosystem services.

The final section of this chapter discusses approaches for assessing trade-offs from management decisions. These approaches aim to quantify, in comparable units, the repercussions of a decision for the full range of ecosystem services. The approaches must also account for the varying spatial and temporal scale over which management decisions alter ecosystem services. Decisions to clear forests, for example, provide immediate economic benefits for local interests but contribute to an increase of greenhouse gases in the atmosphere, with longer-term implications at the global scale.

While this chapter provides a general overview of the available methods and data sources and their applicability to the assessment, individual chapters provide detailed descriptions of data sources used in reference to a particular ecosystem or service. Core data sets used by all chapters to ensure consistency and comparability among the different ecosystems are described in Appendix 2.1.

The data sources and methods used in this report were generally not developed explicitly for this assessment. Yet the combination of approaches—including computer modeling, natural resource and biodiversity inventories, remote sensing and geographic information systems, traditional knowledge, case studies,

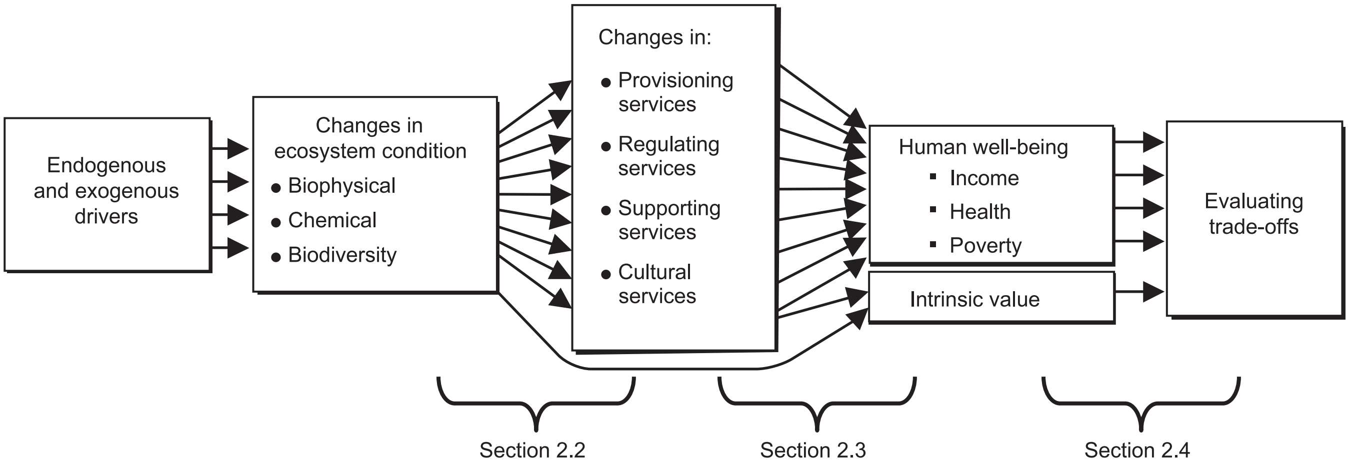

Figure 2.1. Linking Ecosystem Condition to Well-being Requires Assessing Ecosystem Condition and Its Effect on Services, the Impact on Human Well-being and Other Forms of Value, and Trade-offs among Objectives

indicators of ecosystem conditions and human well-being, and economic valuation techniques—provides a strong scientific foundation for the assessment. Systematic data collection for carefully selected indicators reflecting trends in ecosystem condition and their services would provide a basis for future assessments.

2.2 Assessing Ecosystem Condition and Trends

The foundation for analysis is basic information about each ecosystem service (Chapters 7–17) and spatially defined ecosystem (Chapters 18–27). Deriving conclusions about the important trends in ecosystem condition and trade-offs among ecosystem services requires the following basic information:

- What are the current spatial extent and condition of ecosystems?
- What are the quality, quantity, and spatial distributions of services provided by the systems?
- Who lives in the ecosystem and what ecosystem services do they use?
- What are the trends in ecosystem condition and their services in the recent (decades) and more distant past (centuries)?
- How does ecosystem condition, and in turn ecosystem services, respond to the drivers of change for each system?

The availability of data and applicability of methods to derive this basic information (see Table 2.1) vary from ecosystem to ecosystem, service to service, and even region to region within an ecosystem type. For example, the U.N. Food and Agriculture Organization reports data on agricultural products, timber, and fisheries at the national level (e.g., FAO 2000a). Although data reliability is sometimes questionable due to known problems such as definitions that vary between data-submitting countries, data on provisioning ecosystem services with value as commodities are generally available. On the other hand, data on the spatial distribution, quantity, and quality of regulating, supporting, and cultural services such as nutrient cycling, climate regulation, or aesthetic value have generally not been collected, and it is necessary to use indicators, modeled results, or extrapolations from case studies as proxy data. Within a given ecosystem service or geographic system, resource inventories and census data are generally more readily available and reliable in industrial than developing countries.

The following sections provide overviews of each of these data sources and analytical approaches used throughout the report.

2.2.1 Remote Sensing and Geographic Information Systems

The availability of data to monitor ecosystems on a global scale is the underpinning for the MA. Advances in remote sensing technologies over the past few decades now enable repeated observations of Earth's surface. The potential to apply these data for assessing trends in ecosystem condition is only beginning to be realized. Moreover, advances in analytical tools such as geographic information systems allow data on the physical, biological, and socioeconomic characteristics of ecosystems to be assembled and interpreted in a spatial framework, making it feasible to establish linkages between drivers of change and trends in ecosystem services.

2.2.1.1 Remote Sensing

Ground-based surveys for mapping vegetation and other biophysical characteristics can be carried out over limited areas, but it would be an enormous undertaking to carry out globally comprehensive ground-based surveys over the entire surface of Earth. Remote sensing—broadly defined as the science of obtaining information about an object without being in direct physical contact (Colwell 1983)—is the primary data source for mapping the extent and condition of ecosystems over large areas. Moreover, remote sensing provides measurements that are consistent over the entire area being observed and are not subject to varying data collection methods in different locations, unlike ground-based measurements. Repeated observations using the same remote sensing instrument also provide measurements that are consistent through time as well as through space.

Most remote sensing data useful to assess ecosystem conditions and trends are obtained from sensors on satellites. (See Table 2.2.) Satellite data are generally digital and consequently amenable to computer-based analysis for classifying land cover types and assessing trends. There are several types of digital remotely sensed data (Jensen 2000). Optical remote sensing provides digital images of the amount of electromagnetic energy reflected or emitted from Earth's surface at various wavelengths. Active remote sensing of long-wavelengths microwaves (radar), short-wavelength laser light (lidar), or sound waves (sonar) measures the amount of backscatter from electromagnetic energy emitted from the sensor itself.

The spatial resolution (area of ground observed in a picture element or pixel), temporal resolution (how often the sensor re-

Table 2.1. Data Sources and Analytical Approaches for Assessing Ecosystem Condition and Trends

	Data Source or Analytical Method						
Type of Information Required	**Remote Sensing and GIS**	**Natural Resource and Biodiversity Inventories**	**Socioeconomic Data**	**Ecosystem Models**	**Indicators of Ecosystem Condition**	**Indigenous and Traditional Knowledge**	**Case Studies of Ecosystem Response to Drivers**
Current spatial extent and condition of ecosystem	X	X			X		
Quality, quantity, and spatial distributions of services provided by system		X		X			
Human populations residing in and deriving livelihoods from system			X			X	X
Trends in ecosystem conditions and services	X	X		X	X	X	X
Response of ecosystem condition and services to drivers				X	X	X	X

cords imagery from a particular area), spectral resolution (number of specific wavelength intervals in the electromagnetic spectrum to which the sensor is sensitive), and radiometric resolution (precision in the detected signal) determine the utility of the data for a specific application. For example, data with very high spatial resolution can be used to map habitats over local areas, but low temporal resolution limits the ability to map changes over time.

A key element in the interpretation of remote sensing data is calibration and validation with in situ data. Ground-based data aids the interpretation of satellite data by identifying locations of specific features in the land surface. These locations can then be pinpointed on the satellite image to obtain the spectral signatures of different features. Ground-based data are also critical to test the accuracy and reliability of the interpretation of satellite data. Linking ground-based with satellite data poses logistical challenges if the locations required are inaccessible. Moreover, the land surface is often heterogeneous so that a single pixel observed by the satellite contains multiple vegetation types. The ground observations then need to be scaled to the spatial resolution of the sensor. Despite these challenges, ground-based data for calibration and validation are central to the effective use of satellite data for ecosystem assessment.

Analyses of satellite data are a major contribution to assessments of ecosystem conditions and trends, especially over large areas where it is not feasible to perform ground surveys. Technological challenges such as sensor drift and sensor degradation over time, lack of data continuity, and persistent cloud cover, particularly in humid tropics, are challenges to routine application of satellite data to monitor ecosystem condition. Ground observations and local expertise are critical to accurate interpretation of satellite data.

Satellite data contribute to several types of information needs for assessments of ecosystem condition, including land cover and land cover change mapping, habitat mapping for biodiversity, wetland mapping, land degradation assessments, and measurements of land surface attributes as input to ecosystem models.

2.2.1.1.1 Mapping of land cover and land cover change

Over the last few decades, satellite data have increasingly been used to map land cover at national, regional, continental, and global scales. During the 1980s, pioneering research was conducted to map vegetation at continental scales, primarily with data acquired by the U.S. National Oceanographic and Atmospheric Administration's meteorological satellite, the Advanced Very High Resolution Radiometer. Multitemporal data describing seasonal variations in photosynthetic activity were used to map vegetation types in Africa (Tucker 1985) and South America (Townshend 1987). In the 1990s, AVHRR data were used to map land cover globally at increasingly higher spatial resolution, with the first global land cover classification at 1x1 degree resolution (approximately 110x110 kilometers) (DeFries and Townshend 1994), followed by 8x8 kilometer resolution (DeFries 1998) and finally 1x1 kilometer resolution (Loveland and Belward 1997; Hansen 2000).

Global satellite data also have enabled mapping of fractional tree cover to further characterize the distributions of forests over Earth's surface (DeFries 2000). At pantropical scales, AVHRR data have been used to map the distribution of humid forests (Malingreau 1995; Mayaux 1998), and radar data provide useful information for mapping land cover types where frequent cloud cover presents difficulties for optical data (DeGrandi 2000; Saatchi 2000; Mayaux et al. 2002). A suite of recently launched sensors, including MODIS, SPOT Vegetation, and GLI, provide globally comprehensive data to map vegetation types with greater accuracy due to improved spectral, spatial, and radiometric resolutions of these sensors (Friedl 2002). The GLC2000 land cover map derived from SPOT Vegetation data provides the basis for the MA's geographic designation of ecosystems (Bartholome and Belward 2004; Fritz et al. 2004). (See Appendix 2.1.)

One of the most significant contributions to be gained from satellite data is the identification and monitoring of land cover change, an important driver of changes in ecosystem services.

Table 2.2. Satellite Sensors for Monitoring Land Cover, Land Surface Properties, and Land and Marine Productivity

Platform	Sensor	Spatial Resolution at Nadir	Date of Observations
Coarse Resolution Satellite Sensors (> 1 km)			
NOAA–TIROS (National Oceanic and Atmospheric Administration–Television and Infrared Observation Satellite)	AVHRR (Advanced Very High Resolution Radiometer)	1.1km (local area coverage) 8km (global area coverage)	1978–present
SPOT (Systéme Probatoire pour la Observation de la Terre)	VEGETATION	1.15km	1998–present
ADEOS-II (Advanced Earth Observing Satellite)	POLDER (Polarization and Directionality of the Earth's Reflectances)	7km x 6km	2002–present
SeaStar	SeaWIFS (Sea viewing Wide Field of View)	1km (local coverage); 4km (global coverage)	1997–present
Moderate Resolution Satellite Sensors (250 m–1 km)			
ADEOS-II (Advanced Earth Observing Satellite)	GLI (Global Imager)	250m–1km	2002–present
EOS AM and PM (Earth Observing System)	MODIS (Moderate Resolution Spectroradiometer)	250–1,000m	1999–present
EOS AM and PM (Earth Observing System)	MISR (Multi-angle Imaging Spectroradiometer)	275m	1999–present
Envisat	MERIS (Medium Resolution Imaging Spectroradiometer)	350–1,200m	2002–present
Envisat	ASAR (Advanced Synthetic Aperature Radar)	150–1,000m	2002–present
High Resolution Satellite Sensors (20 m–250 m)[a]			
SPOT (Systéme Probatoire pour la Observation de la Terre)	HRV (High Resolution Visible Imaging System)	20m; 10m (panchromatic)	1986–present
ERS (European Remote Sensing Satellite)	SAR (Synthetic Aperture Radar)	30m	1995–present
Radarsat		10–100m	1995–present
Landsat (Land Satellite)	MSS (Multispectral Scanner)	83m	1972–97
Landsat (Land Satellite)	TM (Thematic Mapper)	30m (120m thermal-infrared band)	1984–present
Landsat (Land Satellite)	ETM+ (Enhanced Thematic Mapper)	30m	1999–present
EOS AM and PM (Earth Observing System)	ASTER (Advanced Spaceborne Thermal Emission and Reflection Radiometer)	15–90m	1999–present
IRS (Indian Remote Sensing)	LISS 3 (Linear Imaging Self-scanner)	23m; 5.8m (panchromatic)	1995–present
Very High Resolution Satellite Sensors (< 20 m)[a]			
JERS (Japanese Earth Resources Satellite)	SAR (Synthetic Aperature Radar)	18m	1992–98
JERS (Japanese Earth Resources Satellite)	OPS	18mx24m	1992–98
IKONOS		1m panchromatic; 4m multispectral	1999–present
QuickBird		0.61m panchromatic; 2.44m multispectral	2001–present
SPOT–5	HRG–HRS	10m; 2.5m (panchromatic)	2002–present

Note: The list is not intended to be comprehensive.

[a] Data were not acquired continuously within the time period.

Data acquired by Landsat and SPOT HRV have been the primary sources for identifying land cover change in particular locations. Incomplete spatial coverage, infrequent temporal coverage, and large data volumes have precluded global analysis of land cover change. With the launch of Landsat 7 in April 1999, data are obtained every 16 days for most parts of Earth, yielding more comprehensive coverage than previous Landsat sensors. Time series of Landsat and SPOT imagery have been applied to identify and measure deforestation and regrowth mainly in the humid tropics (Skole and Tucker 1993; FAO 2000a; Achard 2002). Deforestation is the most measured process of land cover change at the regional scale, although major uncertainties exist about absolute area and rates of change (Lepers et al. 2005).

Data continuity is a key requirement for effectively identifying land cover change. With the exception of the coarse resolution AVHRR Global Area Coverage observations over the past 20 years, continuous global coverage has not been possible. DeFries et al. (2002) and Hansen and DeFries (2004) have applied the AVHRR time series to identify changes in forest cover over the last two decades, illustrating the feasibility of using satellite data to detect these changes on a routine basis. Continuity of observations in the future is an essential component for monitoring land cover change and identifying locations with rapid change. For long-term data sets that cover time periods longer than the lifetime of a single sensor, cross calibration for a period of overlap is necessary. Moreover, classification schemes used to interpret the satellite data need to be clearly defined and flexible enough to allow comparisons over time.

2.2.1.1.2 Applications for biodiversity

There are two approaches for applying remote sensing to biodiversity assessments: direct observations of organisms and communities and indirect observations of environmental proxies of biodiversity (Turner et al. 2003). Direct observations of individual organisms, species assemblages, or ecological communities are possible only with hyperspatial, very high resolution (~1m) data. Such data can be applied to identify large organisms over small areas. Airborne observations have been used for censuses of large mammal abundances spanning several decades, for example in Kenya (Broten and Said 1995).

Indirect remote sensing of biodiversity relies on environmental parameters as proxies, such as discrete habitats (for example, woodland, wetland, grassland, or seabed grasses) or primary productivity. This approach has been employed in the US GAP analysis program (Scott and Csuti 1997). Another important indirect use of remote sensing is the detection of habitat loss and fragmentation to estimate the implications for biodiversity based on species-area relationships or other model approaches. (See Chapter 4.)

2.2.1.1.3 Wetland mapping

A wide range of remotely sensed data has been used to map wetland distribution and condition (Darras et al. 1998; Finlayson et al. 1999; Phinn et al. 1999). The utility of such data is a function of spatial and spectral resolutions, and careful choices need to be made when choosing such data (Lowry and Finlayson in press). The NOAA AVHRR, for example, observes at a relatively coarse nominal spatial resolution of 1.1 kilometer and allows only the broad distribution of wetlands to be mapped. More detailed observations of the extent of wetlands can be obtained using finer resolution Landsat TM (30 meters) and SPOT HRV (20 meters) data. As with all optical sensors, the data are frequently affected by atmospheric conditions, especially in tropical coastal areas where humidity is high and the presence of water beneath the vegetation canopy cannot be observed.

Remotely sensed data from newer spaceborne hyperspectral sensors, Synthetic Aperture Radar, and laser altimeters provide more comprehensive data on wetlands. Although useful for providing present-day baselines, however, the historical archive is limited, in contrast to the optical Landsat, AVHRR, and SPOT sensors, which date back to 1972, 1981, and 1986 respectively.

Aerial photographs have been acquired in many years for over half a century at fine spatial resolutions and when cloud cover is minimal. Photographs are available in a range of formats, including panchromatic black and white, near-infrared black and white, true color, and color infrared. Stereo pairs of photographs can be used to assess the vertical structure of vegetation and detect, for example, changes in the extent and height of mangroves (Lucas et al. 2002).

The European Space Agency's project Treaty Enforcement Services using Earth Observation has assessed the use of remote sensing for wetland inventory, assessment, and monitoring using combinations of sensors in support of wetland management. The approach has been extended through the GlobWetland project and its Global Wetland Information Service project to provide remotely sensed products for over 50 wetlands across 21 countries in Africa, Europe, and North and Central America. The project is designed to support on-the-ground implementation of the Ramsar Convention on Wetlands.

2.2.1.1.4 Assessing land degradation in drylands

Interpretation of remotely sensed data to identify land degradation in drylands is difficult because of large variations in vegetation productivity from year-to-year variations in climate. This variability makes it problematic to distinguish trends in land productivity attributable to human factors such as overgrazing, soil salinization, or burning from variations in productivity due to inter-annual climate variability or cyclical drought events (Reynolds and Smith 2002). Land degradation is defined by the Convention to Combat Desertification as "reduction or loss, in arid, semi-arid and dry sub-humid areas, of the biological or economic productivity of rainfed cropland, irrigated cropland, or ranges, pastures, forests, and woodlands resulting from land uses or from a process or combination of processes, including processes arising from human activities and habitation patterns." Quantifying changes in productivity involves an established baseline of land productivity against which changes can be assessed. Such a baseline is often not available. Furthermore, the inherent variability in year-to-year and even decade-to-decade fluctuations complicates the definition of a baseline.

One approach to assess land productivity is through rain-use efficiency, which quantifies net primary production (in units of biomass per unit time per unit area) normalized to the rainfall for that time period (Prince et al. 1990). Rain-use efficiency makes it possible to assess spatial and temporal differences in land productivity without the confounding factor of climate variability. Several models are available to estimate net primary production, as described later, with some using remotely sensed vegetation indices such as the Normalized Difference Vegetation Index (ratio of red to infrared reflectance indicating vegetative activity) as input data for the models. Studies have examined patterns in NDVI, rain-use efficiency, climate, and land use practices to investigate possible trends in land productivity and causal factors (e.g., Prince et al. 1990; Tucker et al. 1991; Nicholson et al. 1998).

The European Space Agency's TESEO project has examined the utility of remote sensing for mapping and monitoring desertification and land degradation in support of the Convention to Combat Desertification (TESEO 2003). Geostationary satellites such as Meteosat operationally provide basic climatological data, which are necessary to estimate rain-use efficiency and distinguish climatic from land use drivers of land degradation. Operational meteorological satellites, most notably the Advanced Very High Resolution Radiometer, have provided the longest continuous record for NDVI from the 1980s to the present. More recently launched sensors such as VEGETATION on-board SPOT and MODIS on-board the Earth Observation System have been designed specifically to monitor vegetation. Satellite data also identify locations of fire events and burn scars to provide information on changes in dryland condition related to changes in fire regime (Giglio et al. 1999). Applications of microwave sensors such as ERS are emerging as possible approaches to map and monitor primary production. Microwave sensors are sensitive to the amount of living aboveground vegetation and moisture content of the upper soil profile and are appropriate for identifying changes in semiarid and arid conditions.

Advancements in the application of remote sensing for mapping and monitoring land degradation involve not just technical issues but institutional issues as well (TESEO 2003). National capacities to use information and technology transfer currently limit the possible applications.

2.2.1.1.5 Measurements of land surface and marine attributes as input to ecosystem models

Satellite data, applied in conjunction with ecosystem models, provide spatially comprehensive estimates of parameters such as evapotranspiration, primary productivity, fraction of solar radiation absorbed by photosynthetic activity, leaf area index, percentage of solar radiation reflected by the surface (albedo) (Myneni 1992; Sellers 1996), ocean chlorophyll (Doney et al. 2003), and species distributions (Raxworthy et al. 2003). These parameters are related to several ecosystem services. For example, a decrease in evapotranspiration from the conversion of part of a forest to an urban system alters the ability of the forest system to regulate climate. A change in primary production relates to the food available for humans and other species. The satellite-derived parameters provide an important means for linking changes in ecosystem condition with implications for their services—for example, linking changes in climate regulation with changes in land and marine surface properties. (See Chapter 13.)

2.2.1.2 Geographic Information Systems

To organize and analyze remote sensing and other types of information in a spatial framework, many chapters in this report rely on geographic information systems. A GIS is a computer system consisting of computer hardware and software for entering, storing, retrieving, transforming, measuring, combining, subsetting, and displaying spatial data that have been digitized and registered to a common coordinate system (Heywood 1998; Johnston 1998). GIS allows disparate data sources to be analyzed spatially. For example, human population density can be overlain with data on net primary productivity or species endemism to identify locations within ecosystems where human demand for ecosystem services may be correlated with changes in ecosystem condition. Locations of roads can be entered into a GIS along with areas of deforestation to examine possible relationships between the two variables. The combination of remote sensing, GIS, and Global Positioning Systems for field validation is powerful for assessing trends in ecosystem condition (Hoffer 1994; ICSU 2002a).

GIS can be used in conjunction with remote sensing to identify land cover change. A common approach is to compare recent and historical high-resolution satellite images (such as Landsat Thematic Mapper). For example, Figure 2.2 illustrates the changes in forest cover between 1992 and 2001 in Mato Grosso, Brazil. Achard et al. (2002) have used this approach in 100 sample sites located in the humid tropical forests to estimate tropical deforestation.

GIS has also been applied in wilderness mapping, also known as ''mapping human impact.'' These exercises estimate human influence through geographic proxies such as human population density, settlements, roads, land use, and other human-made features. All factors are integrated within the GIS and either summed up with equal weights (Sanderson 2002) or weighted according to perceptions of impact (Carver 2002). This exercise has been carried out at regional scales (for example Lesslie and Maslen 1995; Aplet 2000; Fritz 2001) as well as on a global scale (for example, UNEP 2001; Sanderson 2002). Sanderson et al. (2002) used the approach to estimate the 10% wildest areas in each biome of the world. The U.N. Environment Programme's Global Biodiversity (GLOBIO) project uses a similar methodology and examines human influence in relation to indicators of biodiversity (UNEP 2001).

A further application of GIS and remote sensing is to test hypotheses and responses of ecosystem services to future scenarios (Cleland 1994; Wadsworth and Treweek 1999). For example, GIS is used in the MA's sub-global assessment of Southern Africa to predict the degree of fuelwood shortages for the different districts of Northern Sofala Province, Mozambique, in 2030. This is done by using the GIS database showing available fuelwood per district in 1995 and projecting availability in 2030, assuming that the current trend of forest degradation of 0.05 hectares per person per year will continue. This allows identification of districts where fuelwood would be most affected.

GIS is also applicable for assessing relationships between health outcomes and environmental conditions (see Chapter 14) and for mapping risks of vulnerable populations to environmental stressors (see Chapter 6). The spatial displays aim to delineate the places, human groups, and ecosystems that have the highest risk associated with them. Examples include the ''red data'' maps depicting critical environmental situations (Mather and Sdasyuk 1991), maps of ''environmentally endangered areas'' (National Geographic Society 1989), and locations under risk from infrastructure expansion (Laurance et al. 2001), biodiversity loss (Myers et al. 2000), natural hazards, impacts from armed conflicts (Gleditsch et al. 2002), and rapid land cover change (Lepers et al. 2005). The analytical and display capabilities can draw attention to priority areas that require further analysis or urgent attention. Interactive Internet mapping is a promising approach for risk mapping but is currently in its infancy.

2.2.2 Inventories of Ecosystem Components

Inventories provide data on various ecosystem components relevant to this assessment. The most common and thorough types of inventories relate to the amount and distribution of provisioning services such as timber and agricultural products. Species inventories also provide information useful for assessing biodiversity, and demographic data provide essential information on human populations living within the systems.

2.2.2.1 Natural Resource Inventories

Many countries routinely conduct inventories of their natural resources. These generally assess the locations and amounts of

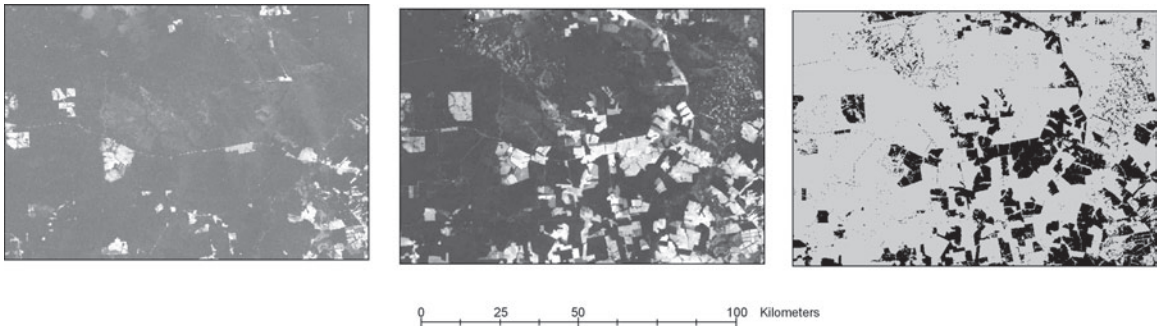

Figure 2.2. Subset of Landsat ETM+ Scenes for an Area in the State of Mato Grosso, Brazil Acquired August 6, 1992 (left) and July 30, 2001 (middle). Light to dark shades represent radiance in band 3 (.63–.62). The difference between the dates indicates deforestation in black (right). The area includes approximately 5534'25"W, 1154'20"S (bottom right corner).

economically important ecosystem services such as timber, agricultural products, and fisheries. FAO periodically publishes compilations of the national-level statistics in forest resources, agricultural production, fisheries production, and water resources. (See Table 2.3.) These statistics are widely used throughout this report. They are in many cases the only source of globally comprehensive data on these ecosystem services. Meta-analyses of local natural resource inventories also provide information on ecosystem condition and trends (Gardner et al. 2003), although they are not spatially comprehensive.

Although the assessment of ecosystem conditions and trends relies heavily on data from resource inventories, there are a number of limitations. First, questions remain about varying methods and definitions used by different countries for data collection (Matthews 2001). For example, several studies based on analysis of satellite data indicate that the FAO Forest Resource Assessment overestimates the rate of deforestation in some countries (Steininger 2001; Achard 2002; DeFries 2002). For fisheries, there are no globally consistent inventories of fisheries and fishery resources. Efforts to develop them are only starting, with the implementation of the FAO Strategy for Improving Information on Status and Trends of Capture Fisheries, which was adopted in 2003 in response to concerns about the reliability of fishery data (FAO 2000b).

Second, resource inventories are often aggregated to the national level or by sub-national administrative units. This level of aggregation does not match the ecosystem boundaries used as the reporting unit for the MA. Third, data quality is highly uneven, with greater reliability in industrial than developing countries. In many countries, deforestation "data" are actually projections based on models rather than empirical observations (Kaimowitz and Angelsen 1998). Fourth, statistics on the production of an ecosystem service do not necessarily provide information about the capacity of the ecosystem to continue to provide the service. For example, fisheries catches can increase for years through "mining" of the stocks even though the underlying biological capability of producing fish is declining, eventually resulting in a collapse. Finally, inventories for noncommodity ecosystem services, particularly the regulating, supporting, and cultural services, have not been systematically carried out.

2.2.2.2 Biodiversity Inventories

Inventories of the biodiversity of ecosystems are far less extensive than those of individual natural resources with value as commodities. Only a small fraction of biodiversity is currently monitored and assessed. This is probably because there are few perceived economic incentives to inventory biodiversity per se and because biodiversity is a complex phenomenon that is difficult to quantify and measure. (See Chapter 4.) Nonetheless, biodiversity inventories can provide a general sense of the relative biodiversity importance (such as richness, endemism) of ecosystems; they can illuminate the impacts of different human activities and management policies on biodiversity; and, when targeted at service-providing taxa or functional groups (pollinators, for instance), they can link changes in biodiversity within these groups directly to changes in the service provided.

Biodiversity inventories are conducted at a range of spatial scales, which are chosen to best address the issue or question at hand. Most, however, can be usefully grouped into three distinct categories: global inventories, regional inventories, and local inventories. Because biodiversity is complex, inventories typically focus on one aspect of biodiversity at a time, such as species richness or habitat diversity. A few examples of inventories at each of these scales illustrate their relative strengths, limitations, and utilities for the MA.

At the global scale, only a handful of biodiversity inventories exist. These typically provide species lists for relatively well-known taxa, based on relatively large spatial units. For example, the World Conservation Monitoring Centre (1992) compiled species inventories of mammals, birds, and swallowtail butterflies for all nations in the world. The World Wild Fund for Nature is conducting an inventory of all vertebrates and plants in each of the world's 867 terrestrial ecoregions (defined by WWF as relatively large units of land or water containing a distinct assemblage of natural communities and species, with boundaries that approximate the original extent of natural communities prior to major land use change).

These inventories are useful for documenting overall patterns of biodiversity on Earth, in order to indicate global priorities for biodiversity conservation or areas of high-expected threat (Sisk et al. 1994; Ceballos and Brown 1995; Dinerstein 1995). Their utility for focused analyses is limited, however, by the coarse units on which they are based and their restriction to mostly vertebrate taxa (which are not often the most important for the provision of ecosystem services).

In addition, the World Conservation Union–IUCN has been producing Red Data Books and Red Lists of Threatened Species since the 1960s. Currently, the IUCN Red List is updated annually (see www.redlist.org). The criteria for listing are transparent and quantitative. The IUCN Red List is global in coverage and is the most comprehensive list of threatened species, with almost all known bird, mammal, and amphibian species evaluated; there are plans for complete coverage of reptiles in the next few years. Data on fish species include FISHBASE (Frose and Pauly 2000), Ceph-

Table 2.3. Examples of Resource Inventories Applicable to Assessing Ecosystem Condition and Trends

Type	Source	Description
Forest Resources		
Forest area and change	FAO, *Global Forest Resources Assessment*	Published every 10 years (1980, 1990, 2000). Provides national and global estimates of total forest area and net changes during the preceding decade, as well as information on plantations, forest ownership, management, and environmental parameters such as forest fires and biomass volumes.
Forest products	FAO, *State of the World's Forests*	Published every two years. Provide summary tables of national and regional production statistics for major categories of industrial roundwood, pulp, and paper.
	ITTO, *Annual Review and Assessment of the World Timber Situation*	Published annually. Tabular databases on volume and value of production, consumption, and trade among ITTO producer and consumer countries. Time series for five years prior to publication.
Wood energy	IEA, *Energy Statistics and Balances of OECD and Non-OECD Countries* (four reports)	Published every two years. IEA data since 1994–95 have covered combustible renewables and waste in national energy balances, including disaggregated data for production and consumption of wood, charcoal, black liquor, and other biomass. Data provided at national and various regional aggregate levels.
Agricultural Resources		
Agricultural land, products, and yields	*FAOSTAT-Agriculture* (data available on-line)	Time series data since 1961 on extent of agricultural land use by country and region, production of primary and processed crops, live animals, primary and processed animal products, imports and exports, food balance sheets, agricultural inputs, and nutritional yield of many agricultural products.
Specific products	Member organizations of the Consultative Group on International Agricultural Research	Issue-specific datasets on crops, animals, animal products, agricultural inputs, and genetic resources. Variety of spatial and temporal scales.
Fish Resources		
Fish stocks	FAO, *Review of the State of World Fishery Resources: Marine Fisheries*	Tabular information on the state of exploitation, total production, and nominal catches by selected species groups for major world fisheries.
Marine and inland fisheries	FAO, *FISHSTAT* (data available on-line at www.fao.org/fi/statist/statist.asp)	Databases on fishery production from marine capture and aquaculture, fish commodity production, and trade. Global, regional, and national data. Time series range from 20 to 50 years.
	FAO, *The State of World Fisheries and Aquaculture*	Published every two years. Data on five-year trends in fisheries production, utilization, and trade for the world and for geographic and economic regions. National data for major fishing countries. Also provides extensive analysis of fishery issues.
	FAO, *Yearbook of Fishery Statistics*	Updated annually. Includes aquaculture production and capture production by country, fishing area, principal producers, and principal species. Also trade data in fishery products.
	FAO, *Fisheries Global Information System,* at www.fao.org/fi/figis	Information on aquatic species, marine fisheries, fisheries issues, and, under development in collaboration with regional fishery bodies, the state of marine resources and inventories of fisheries and fishery resources.
	International Center for Living Aquatic Management, *FishBase 2000*	Database on more than 27,000 fish species and references. Many datasets incomplete.
Freshwater/Inland Water Resources		
Water resources	FAO, *AQUASTAT*	Global data on water resources and irrigation by country and region. Information on average precipitation, total internal water resources, renewable groundwater and surface water, total renewable water resources, and total exploitable water resources.
	State Hydrological Institute (Russia) and UNESCO, *World Water Resources and Their Use,* 1999	Global database on surface water resources and sectoral use. Includes water use forecasts to 2025.

BASE (Wood et al. 2000), ReefBase (Oliver et al.), and the Census of Marine Life (O'Dor 2004). Freshwater fish species are also being evaluated on a region basis for inclusion in the IUCN Red Lists.

Inventories at regional or continental scales are generally of higher overall quality and are more common than global data. Many of these data sets are based on grids of varying resolution. Examples include data on vertebrates in sub-Saharan Africa (grid size 1 degree or approximately 110 square kilometers) (Balmford et al. 2001), birds in the Americas (grid size 611,000 square kilometers) (Blackburn and Gaston 1996), several taxa of plants and animals in Britain (grid size 10 square kilometers) (Prendergast et al. 1993), and terrestrial vertebrates and butterflies in Australia (grid size 1 degree) (Luck et al. 2004). These grid-based inventories, as well as others based on political boundaries (countries, states) are based on arbitrary units that rarely reflect ecosystem boundaries. As a result, their utility is limited in assessing the biodiversity of a particular ecosystem. Some regional-scale inventories are based on ecological units, including a study on vertebrates, butterflies, tiger beetles, and plants for 116 WWF ecoregions in North America (Ricketts et al. 1999).

All these regional inventories can be used to understand patterns of biodiversity and endangerment (e.g., Ceballos and Brown 1995) and to link these patterns to threats and drivers operating at regional scales (e.g., Balmford et al. 2001; Ricketts in press). As is often the case, these data sets are most complete and dependable in the industrial world, although data are improving in many developing regions.

Because many ecosystem services (such as pollination and water purification) are provided locally, local-scale biodiversity inventories are often the most directly valuable for assessing those services. There are thousands of local inventories in the literature, comparing biodiversity between ecosystem types, among land use intensities, and along various environmental gradients. This literature has not been systematically compiled, and it is not possible to list all the studies here.

We illustrate the types of available data here with biodiversity studies in agricultural landscapes dominated by coffee cultivation. Local inventories in these landscapes have quantified the decline in both bird (Greenberg et al. 1997) and arthropod (Perfecto et al. 1997) diversity with increasing intensification of coffee production. Other studies have shown a decline in moth (Ricketts et al. 2001) and bird (Luck and Daily 2003) diversity with increasing distance from remnant patches of forest. Most relevant to ecosystem services supporting coffee production, the diversity and abundance of coffee-visiting bees declines with increasing distance from forest (Ricketts in press) and with increasing intensification (Klein et al. 2002).

Local inventories offer data that can directly inform land use policies and illuminate trade-offs among ecosystem services for decision-makers. Unfortunately, they are often time- and resource-intensive. In addition, the results are only relevant to the specific taxon and location under study, so general lessons are often difficult to glean. However, the collective results of many such studies can lead to useful general guidelines and principles.

Another method of compiling results from many biodiversity inventories is to examine the collections of museums and herbaria (Ponder et al. 2001). These house enormous amounts of information, accumulated sometimes over centuries of study. Furthermore, museums are beginning to use information technologies and the Internet to pool their information into aggregate databases, such that records from any museum can be searched (e.g., Edwards et al. 2000). These aggregate databases are an invaluable resource for studying the distribution of biodiversity. Museum and herbaria records, however, often contain a variety of spatial, temporal, and taxonomic biases and gaps due to the ad hoc and varying interests of collecting scientists (Ponder et al. 2001). These biases must be carefully considered when using museum data to assess biodiversity status and trends.

Ideally, data for characterizing biodiversity in the individual systems and its response to changes in ecosystem condition would be collected routinely according to an appropriate sampling strategy that meets the needs of the specified measures. Most often this is not the case, however, and data assimilated for other purposes are used, such as routine or sporadic surveys and observations made by naturalists. Generally such observations relate only to the most obvious and common species, especially birds and sometimes mammals, butterflies, and so on.

2.2.2.3 Demographic and Socioeconomic Data on Human Populations

Because the MA considers human populations as integral components of ecosystems, data on the populations living within the systems are one of the foundations for this analysis. Demographic and socioeconomic data provide information on the distributions of human populations within ecosystems, a prerequisite to analyzing the dependence of human well-being on ecosystem services.

Most information on the distribution and characteristics of human population is collected through population censuses and surveys. Nearly all countries of the world conduct periodic censuses (see www.census.gov/ipc/www/cendates/cenall.pdf); most countries conduct them once per decade. Census data are collected and reported by administrative or political units, such as counties, provinces, or states. These administrative boundaries generally do not correspond to the geographic boundaries of ecosystems.

To address this mismatch, the most recent version of the Gridded Population of the World (version 3) (CIESIN et al. 2004; CIESIN and CIAT 2004) contains population estimates for over 350,000 administrative units converted to a grid of latitude-longitude quadrilateral cells at a nominal spatial resolution of 5 square kilometers at the equator (Deichmann et al. 2001). The accuracy depends on the quality and year of the input census data and the resolution of the administrative units. Other data sets show how population is distributed relative to urban areas, roads, and other likely population centers, such as LandScan, which uses many types of ancillary data, including land cover, roads, night-time lights, elevation and slope, to reallocate populations within administrative areas to more specific locations (Dobson 2000).

There are large data gaps on poverty distribution and access to ecosystem services such as fresh water (UNDP 2003). Some census data include resource use such as fuelwood and water source (Government of India 2001), but inventories on the use of ecosystems services are not generally available to establish trends. Increasingly, however, censuses and large-scale surveys are beginning to include questions on resource use. The World Bank's Living Standards Measurement Survey, for example, is introducing modules on resource use (Grosh and Glewwe 1995). As most nationally representative socioeconomic and demographic surveys are not georeferenced beyond administrative units, they must be used with care when making inferences at the moderate and high resolutions often used in ecological data analysis.

By combining census information about human settlements with geographic information, such as city night-time lights from satellite data, a new global database indicates urban areas from rural ones (CIESIN et al. 2004). These can be applied to distinguish urban and rural land areas in different ecosystems and to infer implications for resource use. (See Chapter 27.)

2.2.3 Numerical Simulation Models

Numerical models are mathematical expressions of processes operating in the real world. The ecological and human interactions within and among ecosystems are complex, and they involve physical, biological, and socioeconomic processes occurring over a range of temporal and spatial scales. Models are designed as simplified representations to examine assumptions and responses to driving forces.

Models span a wide range in complexity with regard to processes and spatial and temporal scales. Simple correlative models use statistical associations established where data are adequate in order to predict responses where data are lacking. For example, the CLIMEX model (Sutherst 1995) predicts the performance of an insect species in a given location and year in response to climate change based on previously established correlations from comparable locations and previous years. Dynamic, process-based models, on the other hand, are sets of mathematical expressions describing the interactions among components of a system at a specified time step. For example, the CENTURY model simulates fluxes of carbon, water, and nitrogen among plant and soil pools within a grassland ecosystem (Parton 1988). An emerging class of models, such as IBIS (Foley 1996) and LPJ (Sitch et al. 2003), incorporate dynamic processes but also simulate the dynamics of interacting species or plant functional types. Such models have been applied at the site, regional, and global scales to investigate ecosystem responses to climate change scenarios and increasing atmospheric carbon dioxide concentrations (e.g., Cramer et al. 2004).

Table 2.4 lists categories of models useful for the assessment of ecosystem condition and services. These models address various aspects of ecosystem condition. For example, hydrologic models can be used to investigate the effects of land cover changes on flood protection, population models can assess the effects of habitat loss on biodiversity, and integrated assessment models can synthesize this information for assessing effects of policy alternatives on ecosystem condition. Assessments rely on models to:

- **Fill data gaps.** As noted, data to assess trends in ecosystem condition and their services are often inadequate, particularly for regulating, supporting, and cultural services. Models are used to address these deficiencies. For example, Chapter 13 uses results from four ecosystem models (McGuire 2001) to estimate the impacts of changes in land use, climate, and atmospheric composition on carbon dioxide emissions from ecosystems.
- **Quantify responses of ecosystem services to management decisions.** One of the major tasks for the MA is to assess how changes in ecosystem condition alter services. Does removal of forest cover within a watershed alter flood protection? Does conversion to cropland alter climate regulation? Models can be used to simulate changes in the ecosystem condition (such as land cover) and estimate the response (in stream flow, for instance). A hydrologic model (e.g., Liang 1996) can quantify the change in stream flow in response to removal of forest cover. A land surface model linked to a climate model (e.g., Sellers 1986) can quantify the change in water and energy fluxes to the atmosphere from a specified change in land cover and the resulting effect on surface temperature. To the extent that models are adequate representations of reality, they provide an important tool for quantifying the effects of alternative management decisions on ecosystem services.
- **Predict long-term ecological consequences of altered ecosystem condition.** Many human activities affect ecosystem condition only after a time lag. As a consequence, some effects of ecosystem management are not observed for many years. In such cases, models can be used to predict long-term ecological consequences. For example, the effect of timber harvest on the persistence of threatened species such as the spotted owl can be assessed using habitat-based metapopulation models (Akçakaya and Raphael 1998).

  The reliability of long-term model predictions depends on the level of understanding of the system, the amount and quality of available data, the time horizon, and the incorporation of uncertainty. Predictions about simpler systems (such as single-species dynamics) are more reliable than those about complex systems (such as community composition and dynamics), because of the higher level of understanding ecologists have for simpler systems. The amount and quality of the data determine the uncertainty in input parameters, which in turn affect the reliability of the output. Longer-term predictions are less reliable because these uncertainties are compounded over time. Even uncertain predictions can be useful, however, if the level of uncertainty can be objectively quantified. Complex models can also identify shifts in ecosystem regime, such as the sudden loss of submerged vegetation in shallow lakes subject to eutrophication (Scheffer et al. 2001), and nonlinear responses to drivers.
- **Test sensitivities of ecosystem condition to individual drivers or future scenarios.** Observed changes in ecosystem condition result from the combined responses to multiple drivers. Changes in soil fertility in a rangeland, for example, reflect the combined response to grazing pressure, climate variations, and changes in plant species. Direct observations of soil fertility do not enable understanding of which driver is causing the response or how the drivers interact. A series of model simulations, changing one or more drivers for each model run, facilitates understanding of the response of soil fertility to each of the drivers. To the extent that models represent processes realistically, model simulations can identify nonlinear and threshold responses of ecosystems to multiple drivers. For example, neither overfishing nor pollution alone may lead to precipitous declines in fish stocks, but the combined response could have unanticipated effects on fish stocks.
- **Assess future viability of species.** Quantitative methods and models for assessing the chances of persistence of species in the future are collectively called population viability analysis. Models used in PVAs range from unstructured single-population models to metapopulation models with explicit spatial structure based on the distribution of suitable habitat (Boyce 1992; Burgman 1993). PVA provides a rigorous methodology that can use different types of data, incorporate uncertainties and natural variabilities, and make predictions that are relevant to conservation goals. PVA is most useful when its level of detail is consistent with the available data and when it focuses on relative (comparative) rather than absolute results and on risks of decline rather than extinction (Akçakaya and Sjogren-Gulve 2000). An important advantage of PVA is its rigor. In a comprehensive validation study, Brook et al. (2000) found the risk of population decline predicted by PVA closely matched observed outcomes, there was no significant bias, and population size projections did not differ significantly from reality. Further, the predictions of five PVA software packages they tested were highly concordant. PVA results can also be

Table 2.4. Examples of Numerical Models for Assessing Condition and Trends in Ecosystems and Their Services

Type of Model	Description	Examples of Models
Climate and land-atmosphere models	Land surface models of exchanges of water, energy, and momentum between land surface and atmosphere.	Sellers et al. 1986; Liang et al. 1996
Watershed and hydrologic models	Large basin models of hydrologic processes and biogeochemical exchanges in watersheds.	Fekete et al. 2002; Green et al. in press; Seitzinger and Kroeze 1998
Population and metapopulation models	Models of dynamics of single populations predicting future abundance and trends, risk of decline or extinction, and chance of growth. They can be scalar, structured (e.g., age-, stage-, and/or sex-based), or individual-based and incorporate variability, density dependence, and genetics. Metapopulation models focus on the dynamics of and interactions among multiple populations, incorporating spatial structure and dispersal and internal dynamics of each population. Their spatial structure can be based on the distribution and suitability of habitat, and they can be used to assess species extinction risks and recovery chances.	Akçakaya 2002; Lacy 1993
Community or food-web models	Models focusing on the interactions among different trophic levels (producers, herbivores, carnivores) or different species (e.g., predator-prey models).	Park 1998; USDA 1999
Ecosystem process models	Models that include both biotic and abiotic components and that represent physical, chemical, and biological processes in coastal, freshwater, marine, or terrestrial systems. They can predict, for example, vegetation dynamics, including temporal changes in forest species and age structure.	Pastorok et al. 2002
Global terrestrial ecosystem models	Models of biogeochemical cycling of carbon, nitrogen, and other elements between the atmosphere and biosphere at the global scale, including vegetation dynamics, productivity, and response to climate variability.	Field et al. 1995; Foley et al. 1996; McGuire et al. 2001; Sitch et al. 2003
Multi-agent models	Agents are represented by rules for behavior based on interactions with other actors or physical processes.	Moss et al. 2001
Integrated assessment models	Models that assemble, summarize, and interpret information to communicate to decision-makers.	Alcamo et al. 1994

tested for single models by comparing predicted values with those observed or measured in the field (McCarthy 2001).

- **Understand the dynamics of social environmental interactions.** Individually based methods such as multiagent modeling are increasingly used to understand social and environmental interactions. Multiagent behavioral systems seek to model social-environment interactions as dynamic processes (see Moss et al. 2001). Human actors are represented as software agents with rules for their own behavior, interactions with other social agents, and responses to the environment. Physical processes (such as soil erosion) and institutions or organizations (such as an environmental regulator) may also be represented as agents. A multiagent system could represent multiple scales of vulnerability and produce indicators of multiple dimensions of vulnerability for different populations. Multiagent behavioral systems have an intuitive appeal in participatory integrated assessment. Stakeholders may identify with particular agents and be able to validate a model in qualitative ways that is difficult to do for econometric or complex dynamic simulation models. However, such systems require significant computational resources (proportional to the number of agents), and a paucity of data for validation of individual behavior is a constraint.

Models are useful tools for ecosystem assessments if the selection of models, input data, and validation are considered carefully for particular applications. A model developed with data from one location is not directly applicable to other locations. Moreover, data to calibrate and validate models are often difficult to obtain. The appropriateness of a model for an assessment task also depends as much on the capacity of the model variables to capture the values and interests of the decision-making and stakeholding communities as on the accuracy of the underlying scientific data.

2.2.4 Indicators of Ecosystem Condition and Services

An indicator is a scientific construct that uses quantitative data to measure ecosystem condition and services, drivers of changes, and human well-being. Properly constituted, an indicator can convey relevant information to policymakers. In this assessment, indicators serve many purposes, for example:

- as easily measured quantities to serve as surrogates for more difficult to measure characteristics of ecosystem condition—for example, the presence of fecal coliform in a stream is relatively easy to measure and serves a surrogate for poor sanitation in the watershed, which is more difficult to measure.
- as a means to incorporate several measured quantities into a single attribute as an indicator of overall condition—for example, the widely used Index of Biotic Integrity is an indicator of aquatic ecosystem condition (Karr et al. 1986). The IBI is an additive index combining measures of abundances of different taxa. The individual measures can be weighted according to the importance of each taxa for aquatic health.
- as a means to communicate effectively with policy-makers regarding trends in ecosystem conditions and services—for example, information on trends in disease incidence reflects trends in disease control as a ''regulating'' ecosystem service. The former can be readily communicated to a policymaker.
- as a means to measure the effectiveness of policy implementation.

Identifying and quantifying the appropriate indicators is one of the most important aspects of the chapters in this report because it is simply not possible to measure and report all aspects of ecosystems and their relation to human well-being. It is also important to identify appropriate indicators to establish a baseline against which future ecosystem assessments can be compared.

Indicators are designed to communicate information quickly and easily to policy-makers. Economic indicators, such as GDP, are highly influential and well understood by decision-makers. Measures of poverty, life expectancy, and infant mortality directly convey information about human well-being. Some environmental indicators, such as global mean temperature and atmospheric carbon dioxide concentrations, are becoming widely accepted as measures of anthropogenic effects on global climate. Measures of ecosystem condition are far less developed, although some biophysical measures such as spatial extent of an ecosystem and agricultural output are relatively easy to quantify. There are at this time no widely accepted indicators to measure trends in supporting, regulating, or cultural ecosystem services, much less indicators that measure the effect of changes in these services on human well-being. Effective indicators meet a number of criteria (NRC 2000). (See Box 2.1.)

The U.S. National Research Council (NRC 2000) identifies three categories of ecological indicators. First, the extent and status of ecosystems (such as land cover and land use) indicate the coverage of ecosystems and their ecological attributes. Second, ecological capital, further divided into biotic raw material (such as total species diversity) and abiotic raw materials (such as soil nutrients), indicates the amount of resources available for providing services. Finally, indicators of ecological functioning (such as lake trophic status) measure the performance of ecosystems.

Table 2.5 provides examples of three major types of indicators used in this report. (Indicators of human well-being and their utility for measuring how well-being responds to changes in ecosystem services are described later in this chapter.)

- **Indicators of direct drivers of change.** No single indicator represents the totality of the various drivers. Some direct drivers of change (see MA 2003 and Chapter 3) have relatively straightforward indicators, such as fertilizer usage, water consumption, irrigation, and harvests. Indicators for other drivers, including invasion by non-native species, climate change, land cover conversion, and landscape fragmentation, are not as well developed, and data to measure them are not as readily available. Measures such as the per capita "ecological footprint," defined as the area of arable land and aquatic ecosystems required to produce the resources used and assimilate wastes produced per person (Rees 1992), attempt to quantify the demand on ecosystem services into a single indicator. (See Chapter 27.)
- **Indicators of ecosystem condition.** Indicators of biophysical condition of ecosystems do not directly reflect the cause and effect of the drivers but nevertheless can contribute to policy formulation by directing attention to changes of importance. To determine causal relationships, models of interactions among variables must be used. As an analogy with human health, an increase in body temperature indicates infection that warrants further examination. As an example in the biophysical realm, declining trends in fish stocks can trigger investigations of possible causal mechanisms and policy alternatives. Indicators of ecosystem condition include many dimensions, ranging from the extent of the ecosystem to demographic characteristics of human populations to amounts of chemical contaminants (The H. John Heinz III Center for Science, Economics, and the Environment 2002).
- **Indicators of ecosystem services.** Indicators for the provisioning services discussed in Chapters 7–17 generally relate to commodity outputs from the system (such as crop yields or fish) and are readily communicable to policy-makers. Indicators related to the underlying biological capability of the system to maintain the production through supporting and regulating services are a greater challenge. For example, indicators measuring the capability of a system to regulate climate, such as evapotranspiration or albedo, are not as readily interpretable for a policy-maker.

Indicators are essential, but they need to be used with caution (Bossel 1999). Over-reliance on indicators can mask important changes in ecosystem condition. Second, while it is important that indicators are based on measurable quantities, the selection of indicators can be biased toward attributes that are easily quantifiable rather than truly reflective of ecosystem condition. Third, comparing indicators and indices from different temporal and spatial scales is challenging because units of measurement are often inconsistent. Adding up and combining factors has to be done very carefully and it is crucial that the method for combining individual indicators is well understood.

Indicators of biodiversity are particularly important for this assessment. Indicators of the amount and variability of species within a defined area can take many forms. The most common measures are species richness—the number of species—and species diversity, which is the number of species weighted by their relative abundance, biomass, or other characteristic, as in Shannon-Weiner or other similar indices (Rosenzweig 1995).

These two simple measures do not capture many aspects of biodiversity, however. They do not differentiate between native and invasive or introduced species, do not differentiate among species in terms of sensitivity or resilience to change, and do not focus on species that fulfill significant roles in the ecosystem (such as pollinators and decomposers). Moreover, the result depends on the definition of the area and may be scale-dependent. The measures also may not always reflect biodiversity trends accurately. For example, ecosystem degradation by human activities may temporarily increase species richness in the limited area of the impact. Thus refinements of these simple measures provide more insights into the amount of biodiversity. (See Box 2.2.)

Aggregate indicators of trends in species populations such as the Index of Biotic Integrity for aquatic systems (Karr and Dudley 1981) and the Living Planet Index (Loh 2002) use existing data sets to identify overall trends in species abundance and, by implication, the condition of the ecosystems in which they occur. The

BOX 2.1

Criteria for Effective Ecological Indicators (NRC 2000)

- Does the indicator provide information about changes in important processes?
- Is the indicator sensitive enough to detect important changes but not so sensitive that signals are masked by natural variability?
- Can the indicator detect changes at the appropriate temporal and spatial scale without being overwhelmed by variability?
- Is the indicator based on well-understood and generally accepted conceptual models of the system to which it is applied?
- Are reliable data available to assess trends and is data collection a relatively straightforward process?
- Are monitoring systems in place for the underlying data needed to calculate the indicator?
- Can policymakers easily understand the indicator?

Table 2.5. Examples of Indicators to Assess Ecosystem Condition and Trends

Characteristic Described by Indicator	Example of Indicator	Category of Indicator	Availability of Data for Indicator	Units
Direct drivers of change				
Land cover conversion	area undergoing urbanization	ecological state	high	hectares
Invasive species	native vs. non-native species	ecological capital	medium	percent of plant species
Climate change	annual rainfall	ecological state	high	millimeters per year
Irrigation	water usage	ecological functioning	high	cubic meters per year
Ecosystem condition				
Condition of vegetation	landscape fragmentation	ecological state	medium	mean patch size
Condition of soil	soil nutrients	ecological capital	medium	nutrient concentration
	soil salinization	ecological state	low	salt concentration
Condition of biodiversity	species richness	ecological capital	low	number of species/unit area
	threatened species	ecological functioning	medium	percent of species at risk
	visibility of indicator species	ecological functioning	low-medium	probability of extinction
Condition of fresh water	presence of contaminants	ecological state	high	concentration of pollutants index of biotic integrity
Ecosystem service				
Production service	food production	ecological functioning	high	yield (kilograms per hectare per year)
Capacity to mitigate floods	change in stream flow per unit precipitation	ecological capital	low	discharge (cubic meters per second)
Capacity for cultural services	spiritual value	ecological capital	low	?
Capacity to provide biological products	biological products of potential value	ecological capital	low	number of products or economic value

Note: See section 2.3.4 for indicators of human well-being.

BOX 2.2

Indicators of Biodiversity

The following is a sample of the types of indicators that can be used to monitor status and trends in biodiversity. The list is not exhaustive, and specific choice of indicators will depend on particular scale and goals of the monitoring program.

- **Threatened species:** the number of species that are in decline or otherwise classified as under threat of local or global extinction.
- **Indicator species:** species that can be shown to represent the status or diversity of other species in the same ecosystem. Indicator species have been explored as proxies for everything from whole ecosystem restoration (e.g., Carignan and Villard 2002) to overall species richness (e.g., MacNally and Fleishman 2002). The phrase ''indicator species'' is also used broadly to include several of the other categories listed here.
- **Umbrella species:** species whose conservation is expected to confer protection of other species in the same ecosystem (for example, species with large area requirements). If these species persist, it is assumed that others persist as well (Roberge and Angelstam 2004).
- **Taxonomic diversity:** the number of species weighted by their evolutionary distinctiveness (Mace et al. 2003). This indicator is increased with both high species richness and high levels of taxonomic diversity among species. Care is needed that the indicator of taxonomic diversity represents lineage in evolutionary history.
- **Endemism:** the number of species found only in the specific area (e.g., Ricketts in press). Note that this is a scale-dependent measure: as the area assessed increases, higher levels of endemism will result.
- **Ecological role:** species with particular ecological roles, such as pollinators and top predators (e.g., Kremen et al. 2002).
- **Sensitive or sentinel species:** trends in species that react to changes in the environment before other species, especially changes due to human activities (e.g., de Freitas Rebelo et al. 2003). Similar to the famous ''canary in the coal mine,'' monitoring these sensitive species is thought to provide early warning of ecosystem disruption.
- **Aggregate indicators:** indices that combine information about trends in multiple species, such as the Living Planet Index, which aggregates trends in species abundances in forest, fresh water, and marine species (Loh 2002), and the Index of Biotic Integrity, which combines measures of abundances of different taxa in aquatic systems (Karr and Dudley 1981).

Living Planet Index is an aggregation of three separate indices, each the average of trends in species abundances in forest, freshwater, and marine biomes. It can be applied at national, regional, and global levels. The effectiveness of such an aggregate indicator depends on availability and access to data sets on a representative number of species, which is particularly problematic in many developing countries.

The number of species threatened with extinction is an important indicator of biodiversity trends. Using this indicator requires that a number of conditions to be met, however. First, the criteria used to categorize species into threat classes must be objective and transparent and have a scientific basis. Second, the changes in the status of species must reflect genuine changes in the conservation status of the species (rather than changes in knowledge or taxonomy, for example). Third, the pool of species evaluated in two different time periods must be comparable (if more threatened species are evaluated first, the proportion of threatened species may show a spurious decline).

The IUCN Red List of Threatened Species mentioned earlier meets these conditions. The criteria used in assigning species to threat categories (IUCN 2001) is quantitative and transparent yet allows for flexibility and can incorporate data uncertainties (Akçakaya 2000). The IUCN Red List database also records whether or not a species has been evaluated for the first time. For species evaluated previously, the assessment includes reasons for any change in status, such as genuine change in the status of the species, new or better information available, incorrect information used previously, taxonomic change affecting the species, and previous incorrect application of the Red List criteria. Finally, the complete coverage of some taxonomic groups helps make evaluations comparable, although the fact that new species are being evaluated for other groups must be considered when calculating measures such as the proportion of threatened species in those groups.

2.2.5 Indigenous, Traditional, and Local Knowledge

Traditional knowledge broadly represents information from a variety of sources including indigenous peoples, local residents, and traditions. The term indigenous knowledge is also widely used referring to the knowledge held by ethnic minorities from the approximately 300 million indigenous people worldwide (Emery 2000). The International Council for Science defines TK as ''a cumulative body of knowledge, know-how, practices and representation maintained and developed by peoples with extended histories of interaction with the natural environment. These sophisticated sets of understandings, interpretations and meanings are part and parcel of a cultural complex that encompasses language, naming and classification systems, resource use practices, ritual, spirituality and worldview'' (ICSU 2002b).

TK and IK are receiving increased interest as valuable sources of information (Martello 2001) about ecosystem condition, sustainable resource management (Johannes 1998; Berkes 1999; 2002), soil classification (Sandor and Furbee 1996), land use investigations (Zurayk et al. 2001), and the protection of biodiversity (Gadgil et al. 1993). Traditional ecological knowledge is a subset of TK that deals specifically with environmental issues.

Pharmaceutical companies, agribusiness, and environmental biologists have all found TEK to be a rich source of information (Cox 2000; Kimmerer 2000). TEK provides empirical insight into crop domestication, breeding, and management. It is particularly important in the field of conservation biology for developing conservation strategies appropriate to local conditions. TEK is also useful for assessing trends in ecosystem condition (Mauro and Hardinson 2000) and for restoration design (Kimmerer 2000), as it tends to have qualitative information of a single local record over a long time period.

Oral histories can play an important role in the field of vulnerability assessment, as they are especially effective at gathering information on local vulnerabilities over past decades. Qualitative information derived from oral histories can be further developed as storylines for further trends and can lead into role playing simulations of new vulnerabilities or adaptations (Downing et al. 2001).

However, TK has for a long time not been treated equally to knowledge derived from formal science. Although Article 27 of the Universal Declaration of Human Rights of 1948 protects Intellectual Property, the intellectual property rights of indigenous people have often been violated (Cox 2000). The Convention on Biological Diversity of 1992 for the first time established international protocols on the protection and sharing of national biological resources and specifically addressed issues of traditional knowledge. In particular, the parties to the convention agree to respect and preserve TK and to promote wide applications and equitable sharing of its benefits (Antweiler 1998; Cox 2000; Singhal 2000).

The integration of TEK with formal science can provide a number of benefits, particularly in sustainable resource management (Johannes 1998; Berkes 2002). However, integrating TEK with formal science is sometimes problematic (Antweiler 1998; Fabricus et al. 2004). Johnson (1992) cites the following as reasons why integrating TEK is difficult:

- Traditional environmental knowledge is disappearing and there are few resources to document it before it is lost.
- Translating concepts and ideas from cultures based on TEK (mainly oral-based knowledge systems) into the concepts and ideas of formal science is difficult.
- Appropriate methods to document and integrate TEK are lacking, and natural scientists often criticize the lack of rigor of the traditional anthropological methods for interviewing and participant observation
- Integrating TEK and formal science is linked to political power, and TEK is often seen as subordinate.

Moreover, existing practices of TEK, such as forest management, are not necessarily sustainable (Antweiler 1998).

It has been repeatedly pointed out that if TEK is integrated it needs to be understood within its historical, socioeconomic, political, environmental, and cultural location (Berkes 2002). This implies that the ratio of local to scientific knowledge will vary depending on the case and situation (Antweiler 1998). The limitations and shortcomings of integrating TEK and formal science must be addressed, and the methods chosen to collect this knowledge should take the location-specific environments in which they operate into account (Singhal 2000). Integration can also be hindered by different representations of cross-scale interactions, nonlinear feedbacks, and uncertainty in TEK and formal science (Gunderson and Holling 2002). Due to this high degree of uncertainty, it is essential to validate and compare both formal and informal knowledge (Fabricus et al. 2004).

There have been general concerns about scaling up TEK to broader spatial scales, as this traditional knowledge is seldom relevant outside the local context (Forsyth 1999; Lovell et al. 2002). Moreover, analysts warn of a downplaying of environmental problems when TEK is overemphasized. On the other hand, researchers have also warned that efforts to integrate or bridge different knowledge systems will lead inevitably to the compartmentalization and distillation of traditional knowledge into a form that is understandable and usable by scientists and resource managers alone (Nadasdy 1999).

Despite these limitations, TEK—if interpreted carefully and assessed appropriately—can provide important data on ecosystem conditions and trends. The most promising methods of data collection are participatory approaches, in particular Participatory Rural Appraisal (Catley 1996). PRA is an alternative to unstructured visits to communities, which may be biased toward more accessible areas, and to costly, time-consuming questionnaire surveys (Chambers 1994). PRA was developed during the early 1990s from Rapid Rural Appraisal, a cost-effective and rapid way of gathering information. RRA was criticized as being too "quick and dirty" and not sufficiently involved with local people. PRA tries to overcome the criticisms of RRA by allowing recipients more control of problem definition and solution design and by carrying out research over a longer period (Zarafshani 2002; Scoones 1995). Activities such as interviewing, transects, mapping, measuring, analysis, and planning are done jointly with local people (Cornwall and Pratt 2003).

Participatory methods have their limitations: First, they only produce certain types of information, which can be brief and superficial. Second, the information collected may reflect peoples' own priorities and interests. Third, there might be an unequal power relation among participants and between participants and researchers (Cooke and Kothari 2001). Glenn (2003) warns that a rush to obtain traditional knowledge can be biased toward preexisting stereotypes and attention to vocal individuals who do not necessarily reflect consensus.

The MA sub-global assessments used a wide range of participatory research techniques to collect and integrate TEK and local knowledge into the assessment process. In addition to PRA (Pereira 2004), techniques such as focus group workshops (Borrini-Feyerabend 1997), semi-structured interviews with key informants (Pretty 1995), forum theater, free hand and GIS mapping, pie charts, trend lines, timelines, ranking, Venn diagrams, problem trees, pyramids, role playing, and seasonal calendars were used (Borrini-Feyerabend 1997; Jordan and Shrestha 1998; Motteux 2001).

2.2.6 Case Studies of Ecosystem Responses to Drivers

Case studies provide in-depth analyses of responses of ecosystem conditions and services to drivers in particular locations. For example, the study of the Yaqui Valley in Mexico illustrates the response of birds, marine mammals, and fisheries to upland runoff generated by increasing fertilizer use in the heavily irrigated valley (Turner II et al. 2003). Evidence generated from a sufficient number of case studies allows general principles to emerge about ecosystem responses to drivers. Case studies, which can analyze relationships in more detail than would be possible with nationally aggregated statistics or coarse resolution data, also illustrate the range of ecosystem responses to drivers in different locations or under different biophysical conditions.

Few studies have been undertaken to synthesize information from case studies. One such effort analyzed 152 sub-national case studies investigating the response of tropical deforestation to economic, institutional, technological, cultural, and demographic drivers (Geist and Lambin 2001, 2002). The analysis revealed complex relationships between drivers and deforestation in different regions of the tropics, indicating challenges for generic and widely applicable land-use policies to control deforestation. The MA does not carry out such extensive meta-analyses, but rather uses their results where available as well as results from individual case studies from the scientific literature.

Drawing conclusions from case studies must be done with caution. First, individual studies do not generally use standard protocols for data collection and analysis, so comparisons across case studies are difficult. Second, researchers make decisions about where to carry out a case study on an individual basis, so biases might be introduced from inadequate representation from different locations. Third, unless a sufficient number of case studies are available it is not prudent to draw general conclusions and extrapolate results from one location to another. In spite of these limitations, case studies can illustrate possible linkages between ecosystem response and drivers and can fill gaps generated by lack of more comprehensive data when necessary.

2.3 Assessing the Value of Ecosystem Services for Human Well-being

This section addresses the data and methods for assessing the linkages between ecosystem services and human well-being.

2.3.1 Linking Ecosystem Condition and Trends to Well-being

Ecosystem condition is only one of many factors that affect human well-being, making it challenging to assess linkages between them. Health outcomes, for example, are the combined result of ecosystem condition, access to health care, economic status, and myriad other factors. Interpretations of trends in indicators of well-being must appropriately account for the full range of factors involved.

The impacts of ecosystem change on well-being are often subtle, which is not to say unimportant; impacts need not be drastic to be significant. A small increase in food prices resulting from lower yields as a result of land degradation will affect the well-being of many people, even if none starve as a result.

Two basic approaches can be used to trace the linkages between ecosystem condition and trends and human well-being. The first attempts to correlate trends in ecosystem condition to changes in human well-being directly, while the second attempts to trace the impact to the groups affected through biophysical and socioeconomic processes. For example, the impact of water contamination on the incidence of human disease could be estimated by correlating measures of contaminants in water supplies with measures of the incidence of gastrointestinal illnesses in the general population, controlling for other factors that might affect the relationship. Alternatively, the impact could be estimated by using a dose-response function that relates the incidence of illness to the concentration of contaminants to estimate the increase in the probability of illness, then combining that with estimates of the population served by the contaminated water to arrive at a predicted total number of illnesses.

Both approaches face considerable problems. Efforts to correlate ecosystem condition with human well-being directly are difficult because of the presence of multiple confounding factors. Thus the incidence of respiratory illness depends not only on the concentration of airborne contaminants but also on predisposition to illness through factors such as nutritional status or the prevalence of smoking, exposure factors such as the proportion of time spent outdoors, and so on. Analyses linking well-being and ecosystem condition are most easily carried out at a local scale, where the linkages can be most clearly identified.

2.3.2 Measuring Well-being

Human well-being has several key components: the basic material needs for a good life, freedom and choice, health, good social relations, and personal security. Well-being exists on a continuum with poverty, which has been defined as "pronounced deprivation in well-being." One of the key objectives of the MA is to identify the direct and indirect pathways by which ecosystem change can affect human well-being, whether positively or negatively.

Well-being is multidimensional, and so very hard to measure. All available measures have problems, both conceptual (are they measuring the right thing, in the right way?) and practical (how do we actually implement them?). Moreover, most available measures are extremely difficult to relate to ecosystem services.

Economic valuation offers a way both to value a wide range of individual impacts (some quite accurately and reliably, others less so) and, potentially but controversially, to assess well-being as a whole by expressing the disparate components of well-being in a single unit (typically a monetary unit). It has the advantage that impacts denominated in monetary units are readily intelligible and comparable to other benefits or to the costs of intervention. It can also be used to provide information to examine distributional, equity, and intergenerational aspects. Economic valuation techniques are described in the next section.

Health indicators address a key subset of impacts of ecosystem services on well-being. They are an important complement to economic valuation because they concern impacts that are very difficult and controversial to value. Some health indicators address specific types of health impacts; others attempt to aggregate a number of health impacts. Likewise, poverty indicators measure a dimension of well-being that is often of particular interest. These, too, are described later in the chapter.

Numerous other well-being indicators (such as the Human Development Index) have been developed in an effort to capture the multidimensionality of well-being into a single number, with varying degrees of success. Although these indicators are arguably better measures of well-being, they tend not to be very useful for assessing the impact of ecosystems, as many of the dimensions they add (literacy, for instance) tend not to be sensitive to ecosystem condition. These aggregate indicators and the limitations they face are described near the end of this chapter.

2.3.3 Economic Valuation

One of the main reasons we worry about the loss of ecosystems is that they provide valuable services—services that may be lost or diminished as ecosystems degrade. The question then immediately arises: how valuable are these services? Or, put another way, how much worse off would we be if we had less of these services? We need to be able to answer these questions to inform the choices we make in how to manage ecosystems.

Economic valuation attempts to answer these questions. It is based on the fact that human beings derive benefit (or "utility") from the use of ecosystem services either directly or indirectly, whether currently or in the future, and that they are willing to "trade" or exchange something for maintaining these services. As utility cannot be measured directly, economic valuation techniques are based on observation of market and nonmarket exchange processes. Economic valuation usually attempts to measure all services in monetary terms, in order to provide a common metric in which to express the benefits of the diverse variety of services provided by ecosystems. This explicitly does not mean that only services that generate monetary benefits are taken into consideration in the valuation process. On the contrary, the essence of most work on valuation of environmental and natural resources has been to find ways to measure benefits that do not enter markets and so have no directly observable monetary benefits. The concept of Total Economic Value is a framework widely used to disaggregate the utilitarian value of ecosystems into components (Pearce 1993). (See Box 2.3.)

Valuation can be used in many different ways (Pagiola et al. 2004). The MA uses valuation primarily to evaluate trade-offs between alternative ecosystem management regimes that alter the use of ecosystems and the multiple services they provide. This approach focuses on assessing the value of changes in ecosystem services resulting from management decisions or other human ac-

BOX 2.3

Total Economic Value

The concept of total economic value is widely used by economists (Pearce and Warford 1993). This framework typically disaggregates the utilitarian value of ecosystems into direct and indirect use values and non-use values:

- **Direct use values** are derived from ecosystem services that are used directly by humans. They include the value of *consumptive uses,* such as harvesting of food products, timber for fuel or construction, medicinal products, and hunting of animals for consumption, and of *non-consumptive uses,* such as the enjoyment of recreational and cultural amenities like wildlife and bird watching, water sports, and spiritual and social utilities that do not require harvesting of products. Direct use values correspond broadly to the MA notion of provisioning and cultural services. They are typically enjoyed by people located in the ecosystem itself.
- **Indirect use values** are derived from ecosystem services that provide benefits outside the ecosystem itself. Examples include the natural water filtration function of wetlands, which often benefits people far downstream; the storm protection function of coastal mangrove forests, which benefits coastal properties and infrastructure; and carbon sequestration, which benefits the entire global community by abating climate change. This category of benefits corresponds broadly to the MA notion of regulating and supporting services.
- **Option values** are derived from preserving the option to use in the future services that may not be used at present, either by oneself (*option value*) or by others or heirs (*bequest value*). Provisioning, regulating, and cultural services may all form part of option value to the extent that they are not used now but may be used in the future.
- **Non-use values** refer to the value people may have for knowing that a resource exists even if they never use that resource directly. This kind of value is usually known as *existence value* (or, sometimes, *passive use value*). This is one area of partial overlap with non-utilitarian sources of value (see the section on intrinsic value).

The TEV framework does not have any direct analog to the MA notion of supporting services of ecosystems. Rather, these services are valued indirectly, through their role in enabling the ecosystem to provide provisioning and enriching services.

Valuation is usually relatively simple in the case of direct use value, and then increasingly difficult as one moves on to indirect use value, option value, and non-use value.

tions. This type of valuation is most likely to be directly policy-relevant.

Economic valuation has also been used to derive the total value of ecosystem services at a given time (e.g., Costanza et al. 1997) and to simulate the value of ecosystem services in an integrated Earth system model (Boumans et al. 2002). Efforts to estimate the total value of the services being provided by ecosystems at any one time, if conducted properly, can provide useful information on their contribution to economic activity and to well-being. Their usefulness for policy is limited, however, as it is rare for all ecosystem services to be completely lost (and even then, this would usually only happen over time). This chapter, therefore, focuses on methods useful for assessing changes in ecosystem services. (For further discussion of the difference between these approaches, see Bockstael et al. 2000 and Pagiola et al. 2004.)

2.3.3.1 Valuation Methods

Many methods for measuring the utilitarian values of ecosystem services are found in the resource and environmental economics literature (Mäler and Wyzga 1976; Freeman 1979; Hufschmidt et al. 1983; Mitchell and Carson 1989; Pearce and Markandya 1989; Braden and Kolstad 1991; Hanemann 1992; Freeman 1993; Pearce 1993; Dixon et al. 1994; Johansson 1994; Pearce and Moran 1994; Barbier et al. 1995; Willis and Corkindale 1995; Seroa da Motta 1998; Garrod and Willis 1999; Seroa da Motta 2001; Pearce et al. 2002; Turner et al. 2002; Pagiola et al. in review). Table 2.6 summarizes the main economic valuation techniques.

Some techniques are based on actual observed behavior data, including some methods that deduce values indirectly from behavior in surrogate markets, which are hypothesized to have a direct relationship with the ecosystem service of interest. Other techniques are based on hypothetical rather than actual behavior data, where people's responses to questions describing hypothetical markets or situations are used to infer value. These are generally known as ''stated preference'' techniques, in contrast to those based on behavior, which are known as ''revealed preference'' techniques. Some techniques are broadly applicable, some are applicable to specific issues, and some are tailored to particular data sources. As in the case of private market goods, a common feature of all methods of economic valuation of ecosystem services is that they are founded in the theoretical axioms and principles of welfare economics. These measures of change in well-being are reflected in people's willingness to pay or willingness to accept compensation for changes in their level of use of a particular service or bundle of services (Hanemann 1991; Shogren and Hayes 1997). These approaches have been used extensively in recent years, in a wide range of policy-relevant contexts.

A number of factors and conditions determine the choice of specific measurement methods. For instance, when the ecosystem service in question is privately owned and traded in the market, its users have the opportunity to reveal their preferences for that service compared with other substitutes or complementary commodities through their actual market choices, given relative prices and other economic factors. For this group of ecosystem services a demand curve can be derived from observed market behavior, and this allows changes in well-being to be estimated. However, many ecosystem services are not privately owned and not traded, and hence their demand curves cannot be directly observed and measured. Alternative methods have been used to derive values for such ecosystem services.

Valuation is a two-step process. First, the services being valued have to be identified. This includes understanding the nature of the services and their magnitude, and how they would change if the ecosystem changed; knowing who makes use of the services, in what way and for what purpose, and what alternatives they have; and establishing what trade-offs might exist between different kinds of services an ecosystem might provide. The bulk of the work involved in valuation actually concerns quantifying the biophysical relationships. In many cases, this requires tracing through and quantifying a chain of causality. (See Figure 2.3 for an example.) Valuation in the narrow sense only enters in the second step in the process, in which the value of the impacts is estimated in monetary terms.

2.3.3.1.1 Changes in productivity

The most widely used technique, thanks to its broad applicability and its flexibility in using a variety of data sources, is known as the change in productivity technique. It consists of tracing through chains of causality (such as those illustrated in Figure 2.3) so that the impact of changes in the condition of an ecosystem can be related to various measures of human well-being. Such impacts are often reflected in goods or services that contribute directly to human well-being (such as production of crops or of clean water), and as such are often relatively easily valued. The valuation step itself depends on the type of impact but is often straightforward:

- The net value in reductions in irrigated crop production resulting from reduced water availability is easy to estimate, for example, as crops are often sold. (Even so, it is a very common error to use the reduction in the gross value of crop production rather than the net value. Using gross value omits the costs of production and so overestimates the impact.)
- Where the impact is on a good or service that is not marketed or where observed prices are unreliable indicators of value, the valuation can become more complex. The impact of hydrological changes on use of water for human consumption, for example, once again begins by tracing through chains of causality to estimate the changes in the quantity and quality of water available to consumers. This is itself often difficult. The prices typically charged to consumers for this water, moreover, are not reliable measures of the value of the water to consumers, as they are set administratively, with no regard for supply and demand (indeed, in most cases water fees do not even cover the cost of delivering the water to consumers, let alone the value of the water itself). The value of an additional unit of water can be estimated in various ways, such as the cost of alternative sources of supply (cost-based measures are described later) or asking consumers directly how much they would be willing to pay for it (contingent valuation, described later). Note that it is very important to use the value of an additional unit of water, since some amount of water is, of course, vital for survival. Thus an additional unit of water will be very valuable when water is scarce, but much less so when water is plentiful. In this case, as in many others, averages can be misleading.
- When the impact is on water quality rather than quantity, the impact on well-being might be reflected in increased morbidity or even mortality. Again, the process begins by tracing through chains of causality, for example by using dose-response functions that tie concentrations of pollutants to human health. Valuing the impact on health itself can then be done in a number of ways (see cost of illness and human capital, in the next section).
- In some cases, the impact is on relatively intangible aspects of well-being, such as aesthetic benefits or existence value.

Table 2.6. Main Economic Valuation Techniques (Adapted from Pagiola et al. forthcoming)

Methodology	Approach	Applications	Data Requirements	Limitations
Revealed preference methods				
Change in productivity	trace impact of change in environmental services on produced goods	any impact that affects produced goods	change in service; impact on production; net value of produced goods	data on change in service and consequent impact on production often lacking
Cost of illness, human capital	trace impact of change in environmental services on morbidity and mortality	any impact that affects health (e.g., air or water pollution)	change in service; impact on health (dose-response functions); cost of illness or value of life	dose-response functions linking environmental conditions to health often lacking; underestimates, as it omits preferences for health; value of life cannot be estimated
Replacement cost (and variants, such as relocation cost)	use cost of replacing the lost good or service	any loss of goods or services	extent of loss of goods or services; cost of replacing them	tends to overestimate actual value
Travel cost method	derive demand curve from data on actual travel costs	recreation	survey to collect monetary and time costs of travel to destination; distance traveled	limited to recreational benefits; hard to use when trips are to multiple destinations
Hedonic prices	extract effect of environmental factors on price of goods that include those factors	air quality, scenic beauty, cultural benefits	prices and characteristics of goods	requires vast quantities of data; very sensitive to specification
Stated preference methods				
Contingent valuation (CV)	ask respondents directly their willingness to pay for a specified service	any service	survey that presents scenario and elicits willingness to pay for specified service	many potential sources of bias in responses; guidelines exist for reliable application
Choice modeling	ask respondents to choose their preferred option from a set of alternatives with particular attributes	any service	survey of respondents	similar to CV; analysis of the data generated is complex
Other methods				
Benefits transfer	use results obtained in one context in a different context	any for which suitable comparison studies are available	valuation exercises at another, similar site	can be widly inaccurate, as many factors vary even when contexts seem "similar"

Particular efforts have been made in recent years to develop techniques to value such impacts, including hedonic price, travel cost, and contingent valuation methods.

2.3.3.1.2 Cost of illness and human capital

The economic costs of an increase in morbidity due to increased pollution levels can be estimated using information on various costs associated with the increase: any loss of earnings resulting from illness; medical costs such as for doctors, hospital visits or stays, and medication; and other related out-of-pocket expenses. The estimates obtained in this manner are interpreted as lower-bound estimates of the presumed costs or benefits of actions that result in changes in the level of morbidity, since this method disregards the affected individuals' preference for health versus illness and restrictions on non-work activities. Also, the method assumes that individuals treat health as exogenous and does not recognize that individuals may undertake defensive actions (such as using special air or water filtration systems to reduce exposure to pollution) and incur costs to reduce health risks.

When this approach is extended to estimate the costs associated with pollution-related mortality (death), it is referred to as the human-capital approach. It is similar to the change-in-productivity approach in that it is based on a damage function relating pollution to productivity, except that in this case the loss in productivity is that of human beings, measured in terms of expected lifetime earnings. Because it reduces the value of life to the present value of an individual's future income stream, the human-capital approach is extremely controversial when applied to mortality. Many economists prefer, therefore, not to use this approach and to simply measure the changes in the number of deaths (without monetary values) or measures such as disability-adjusted life years (described later).

2.3.3.1.3 Cost-based approaches

The cost of replacing the services provided by the environmental resource can provide an order of magnitude estimate of the value of that resource. For example, if ecosystem change reduces the water filtration services, the cost of treating water to make it meet the required quality standards could be used. The major underlying assumptions of these approaches are that the nature and extent of physical damage expected is predictable (there is an accurate damage function available) and that the costs to replace or restore

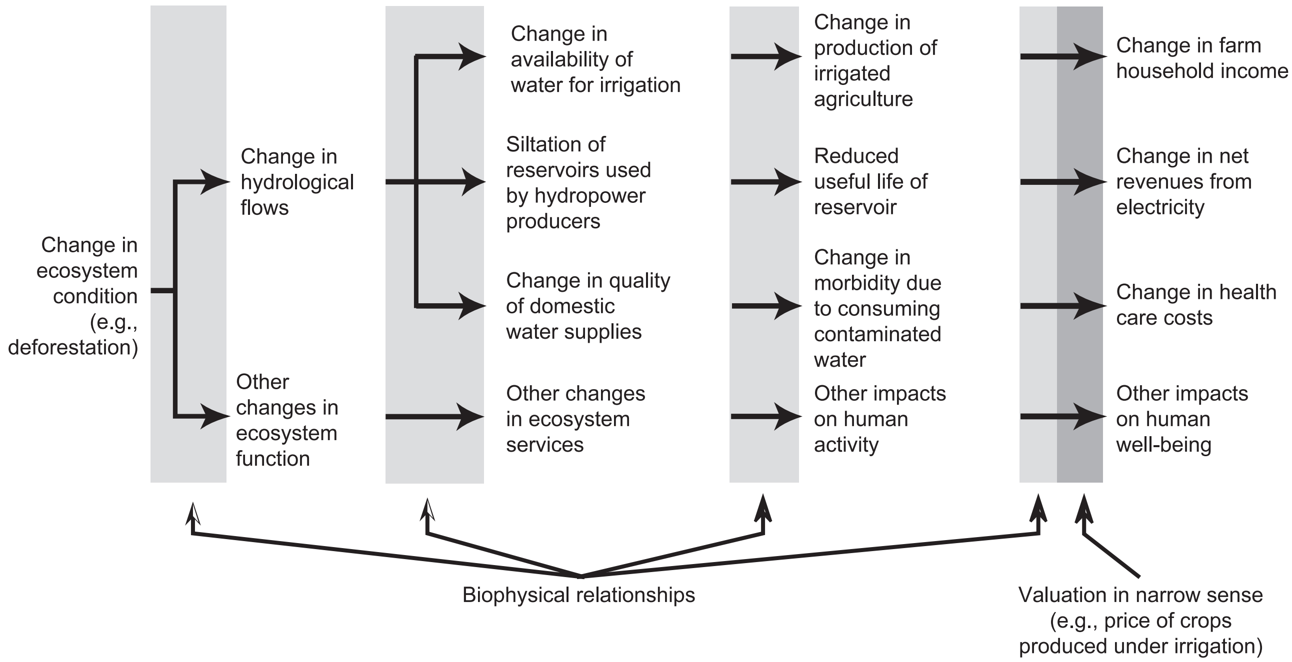

Figure 2.3. Valuing the Impact of Ecosystem Change (Adapted from Pagiola et al. forthcoming)

damaged assets can be estimated with a reasonable degree of accuracy. It is further assumed that the replacement or restoration costs do not to exceed the economic value of the service. This assumption may not be valid in all cases. It simply may cost more to replace or restore a service than it was worth in the first place—for example, because there are few users or because their use of the service was in low-value activities.

As there are often multiple ways that replacement costs could be estimated (for example, the value of lost water filtration services could be estimated based on the cost of restoring the wetland that had provided the service, the cost of treating water to meet quality standards, or the cost of obtaining suitable water from another source), the cheapest option should be considered as the replacement cost estimate. Because of these problems, cost-based approaches are generally thought to provide an upper-bound estimate of value.

2.3.3.1.4 Hedonic analysis

The prices paid for goods or services that have environmental attributes differ depending on those attributes. Thus, a house in a clean environment will sell for more than an otherwise identical house in a polluted neighborhood. Hedonic price analysis compares the prices of similar goods to extract the implicit value that buyers place on the environmental attributes. This method assumes that markets work reasonably well, and it would not be applicable where markets are distorted by policy or market failures. Moreover, this method requires a very large number of observations, so its applicability is limited.

2.3.3.1.5 Travel cost

The travel cost method is an example of a technique that attempts to deduce value from observed behavior in a surrogate market. It uses information on visitors' total expenditure to visit a site to derive their demand curve for the site's services. The technique assumes that changes in total travel costs are equivalent to changes in admission fees. From this demand curve, the total benefit visitors obtain can be calculated. (It is important to note that the value of the site is not given by the total travel cost; this information is only used to derive the demand curve.) This method was designed for and has been used extensively to value the benefits of recreation, but it has limited utility in other settings.

2.3.3.1.6 Contingent valuation

Contingent valuation is an example of a stated preference technique. It is carried out by asking consumers directly about their willingness-to-pay to obtain an environmental service. A detailed description of the service involved is provided, along with details about how it will be provided. The actual valuation can be obtained in a number of ways, such as asking respondents to name a figure, having them choose from a number of options, or asking them whether they would pay a specific amount (in which case, follow-up questions with higher or lower amounts are often used).

CV can, in principle, be used to value any environmental benefit simply by phrasing the question appropriately. Moreover, since it is not limited to deducing preferences from available data, it can be targeted quite accurately to ask about the specific changes in benefits that the change in ecosystem condition would cause. Because of the need to describe in detail the good being valued, interviews in CV surveys are often quite time-consuming. It is also very important that the questionnaire be extensively pre-tested to avoid various sources of bias.

CV methods have been the subject of severe criticism by some analysts. A ''blue-ribbon'' panel was organized by the U.S. Department of Interior following controversy over the use of CV to value damages from the 1989 Exxon Valdez oil spill. The report of this panel (NOAA 1993) concluded that CV can provide useful and reliable information when used carefully, and it provided guidance on doing so. This report is generally regarded as authoritative on appropriate use of the technique.

2.3.3.1.7 Choice modeling

Choice modeling (also referred to as contingent choice, choice experiments, conjoint analysis, or attribute-based stated choice method) is a newer approach to obtaining stated preferences. It consists of asking respondents to choose their preferred option from a set of alternatives where the alternatives are defined by attributes (including the price or payment). The alternatives are designed so that the respondent choice reveals the marginal rate of substitution between the attributes and money. These approaches are useful in cases in which the investigator is interested in the valuation of the attributes of the situation or when the decision lends itself to respondents choosing from a set of alternatives described by attributes.

Choice modeling has several advantages: the control of the stimuli is in the experimenter's hand, as opposed to the low level of control generated by real market data; the control of the design yields greater statistical efficiency; the attribute range can be wider than found in market data; and the introduction or removal of products, services and attributes is easily accomplished (Louviere et al. 2000; Holmes and Adamowicz 2003; Bateman et al. 2004). The disadvantages associated with the technique are that the responses are hypothetical and therefore suffer from problems of hypothetical bias (similar to contingent valuation) and that the choices can be quite complex when there are many attributes and alternatives. The econometric analysis of the data generated by choice modeling is also fairly complex.

2.3.3.1.8 Benefits transfer

A final category of approach is known as benefits transfer. This is not a methodology per se but rather refers to the use of estimates obtained (by whatever method) in one context to estimate values in a different context. For example, an estimate of the benefit obtained by tourists viewing wildlife in one park might be used to estimate the benefit obtained from viewing wildlife in a different park. Alternatively, the relationship used to estimate the benefits in one case might be applied in another, in conjunction with some data from the site of interest (''benefit function transfer''). For example, a relationship that estimates tourist benefits in one park, based in part on their attributes such as income or national origin, could be used in another park, but with data on income and national origin of that park's visitors.

Benefits transfer has been the subject of considerable controversy in the economics literature, as it has often been used inappropriately. A consensus seems to be emerging that benefit transfer can provide valid and reliable estimates under certain conditions. These conditions include the requirement that the commodity or service being valued be very similar at the site where the estimates were made and the site where they are applied and that the populations affected have very similar characteristics. Of course, the original estimates being transferred must themselves be reliable in order for any attempt at transfer to be meaningful.

2.3.3.1.9 Summary of valuation methods

Each of these approaches has seen extensive use in recent years, and considerable literature exists on their application. These techniques can and have been applied to a very wide range of issues (McCracken and Abaza 2001), including the benefits of ecosystems such as forests (Bishop 1999; Kumari 1995; Pearce et al. 2002; Merlo and Croitoru in press), wetlands (Barbier et al. 1997; Heimlich et al. 1998), and watersheds (Aylward 2004; Kaiser and Roumasset 2002). Other studies have focused on the value of particular ecosystems services such as water (Young and Haveman 1985), non-timber forest benefits (Lampietti and Dixon 1995; Bishop 1998), recreation (Bockstael et al. 1991; Mantua at al. 2001; Herriges and Kling 1999), landscape (Garrod and Willis 1992; Powe et al. 1995), biodiversity for medicinal or industrial uses (Simpson et al. 1994; Barbier and Aylward 1996), natural crop pollination (Ricketts in press), and cultural benefits (Pagiola 1996; Navrud and Ready 2002). Many valuation studies are cataloged in the Environmental Valuation Reference Inventory Web site maintained by Environment Canada (EVRI 2004).

In general, measures based on observed behavior are preferred to measures based on hypothetical behavior, and more direct measures are preferred to indirect measures. However, the choice of valuation technique in any given instance will be dictated by the characteristics of the case and by data availability. Several techniques have been specifically developed to cater to the characteristics of particular problems. The travel cost method, for example, was specifically developed to measure the utility derived by visitors to sites such as protected areas and is of limited applicability outside that particular case. The change in productivity approach, on the other hand, is very broadly applicable to a wide range of issues. Contingent valuation is potentially applicable to any issue, simply by phrasing the questions appropriately and as such has become very widely used—probably excessively so, as it is easy to misapply and, being based on hypothetical behavior, is inherently less reliable than measures based on observed behavior. For some types of value, however, stated preference methods may be the only alternative. Thus, existence value can only be measured by stated preference techniques.

In some cases, the value of a given benefit can be estimated in several ways. For example, the value of water purification might be estimated by the avoided health impacts (an application of change in productivity), by the avoided costs of treating water (an application of replacement costs), or by asking consumers for their willingness to pay for clean water (an application of contingent valuation). In such cases, it is appropriate to take the lowest figure as the estimate of the value of the benefit. It would make little sense to consider water purification to be worth 100 based (for example) on willingness to pay if treating the water to achieve the same result would only cost 10.

2.3.3.2 Putting Economic Valuation into Practice

Whatever valuation method is used, framing the question to be answered appropriately is critical. In most policy-relevant cases, the concern is over changes in the level and mix of services provided by an ecosystem. At any given time, an ecosystem provides a specific ''flow'' of services, depending on the type of ecosystem, its condition (the ''stock'' of the resource), how it is managed, and its socioeconomic context. A change in management (whether negative, such as deforestation, or positive, such as an improvement in logging practices) will change the condition of the ecosystem and hence the flow of benefits it is capable of generating. It is rare for all ecosystem services to be lost entirely; a forested watershed that is logged and converted to agriculture, for example, still provides a mix of provisioning, regulating, supporting, and cultural services, even though both the mix and the magnitude of specific services will have changed.

The typical question being asked, then, is whether the total value of the mix of services provided by an ecosystem managed in one way is greater or smaller than the total value of the mix provided by that ecosystem if it were managed in another way. Consequently, an assessment of this change in the value is typically most relevant to decision-makers. Where the change does involve the complete elimination of ecosystem services, such as the conversion of an ecosystem through urban expansion or road-building, then the change in value would equal the total economic value of the services provided by the ecosystem. Measurements of this total value can also be useful to policy-makers as an economic indicator, just as measures of gross domestic product or genuine savings provide policy-relevant information on the state of the economy.

Assessing the change in value of the ecosystem services caused by a management change can be achieved either by explicitly estimating the change in value or by separately estimating the value of ecosystem services under the current and the alternative management regime and then comparing them. If the loss of a given service is irreversible, then the loss of the option of using that service in the future ("option value") should also be included. An important caveat here is that the appropriate comparison is between the ecosystem with and without the management change; this is not the same as a comparison of the ecosystem before and after the management change, as many other factors will typically also have changed.

The actual change in the value of the benefits can be expressed either as a change in the value of the annual flow of benefits, if these flows are relatively constant, or as a change in the value of all future flows. The latter is equivalent to the change in the capital value of the ecosystem and is particularly useful when future flows are likely to vary substantially over time. (It is important to bear in mind that the capital value of the ecosystem is not separate and additional to the value of the flows of benefits it generates; rather, the two are intimately linked in that the capital value is the value of all future flows of benefits.)

Estimating the change in the value of the flow of benefits provided by an ecosystem begins by estimating the change in the physical flow of benefits. This is illustrated in Figure 2.3 for a hypothetical case of deforestation that affects the water services provided by a forest ecosystem. As noted earlier, the bulk of the work involves quantifying the biophysical relationships. Thus, valuing the change in production of irrigated agriculture resulting from deforestation requires estimating the impact of deforestation on hydrological flows, determining how changes in water flows affect the availability of water to irrigation, and then estimating how changes in water availability affect agricultural production. Only at the end of this chain does valuation in the strict sense occur—in putting a value on the change in agricultural production, which in this instance is likely to be quite simple, as it is based on observed prices of crops and agricultural inputs. The change in value resulting from deforestation then requires summing across all the impacts.

Clearly, tracing through these chains requires close collaboration between experts in different disciplines—in the deforestation example, between foresters, hydrologists, water engineers, and agronomists as well as economists. It is a common problem in valuation that information is only available on some links in the chain and often in incompatible units. An increased awareness by the various disciplines involved of what is needed to ensure that their work can be combined with that of others would facilitate more thorough analysis of such issues, including valuation.

In bringing the various strands of the analysis together, there are many possible pitfalls to be wary of. Inevitably, some ecosystem benefits will prove impossible to estimate using any of the available techniques, either because of lack of data or because of the difficulty of extracting the desired information from them. To this extent, estimates of value will be underestimates. Conversely, there is an opposite danger that benefits (even if accurately measured) might be double-counted.

As needed, the analysis can be carried out either from the perspective of society as a whole or from the perspective of individual groups within society. When the analysis is undertaken from the societal perspective, it should include all costs and benefits associated with ecosystem management decisions, which should be valued at their opportunity cost to society (sometimes known as "shadow prices"). In contrast, focusing on a particular group usually requires focusing on a subset of the benefits provided by an ecosystem, as that group may receive some benefits but not others; groups located within an ecosystem, for example, typically benefit most from provisioning services but little from regulating services, whereas downstream users receive few benefits from provisioning services but many benefits from regulating services. It also requires using estimates of value specific to that group (the value of additional water, for example, will be different depending on whether it is used for human consumption or irrigation). The analysis can thus allow for distributional impacts and equity considerations to be taken into account, as well as overall impacts on well-being at the societal level.

This type of disaggregation is also very useful for understanding the incentives that particular groups face in making their ecosystem management decisions. Many ecosystems are mismanaged, from a social perspective, precisely because most groups that make decisions about ecosystem management perceive only a subset of the benefits it provides (Pagiola and Dixon 2001). Understanding how the benefits and costs of ecosystem management are distributed across different groups can also help design mechanisms to align their incentives with those of society (Pagiola and Platais in press).

Assessing the impact of ecosystem change almost always requires comparing costs and benefits at different times. In economic analysis, this is achieved by discounting future costs and benefits so that all are expressed in today's monetary units (Portney and Weyant 1999). Because discounting makes future benefits appear smaller, this practice has been controversial, and some have called for use of lower (perhaps even zero) discount rate when assessing environmental issues. Discount rates, however, reflect preferences for current as opposed to future consumption. Whatever discount rate is chosen, it should be applied in all evaluations involving choices between outcomes occurring at different times.

Similarly, estimating the impact of changes in management on future flows of benefits allows for intergenerational considerations to be taken into account. Here, too, the bulk of the work involved concerns predicting the change in future physical flows; the actual valuation in the narrow sense forms only a small part of the work. Predicting the value that future generations will place on a given service is obviously difficult. Technical, cultural, or other changes could result in the value currently placed on a service either increasing or decreasing. Often, the best that can be done is to simply assume that current values will remain unchanged. If trends suggest that a particular change in values will occur, that can be easily included in the analysis. Such predictions are notoriously unreliable, however.

2.3.4 Indicators of Specific Dimensions of Well-being

Well-being cannot be measured solely in terms of income, nor can non-income aspects of well-being always be expressed in monetary terms. This section reviews several indicators that seek to capture specific aspects of well-being which economic valuation often captures imperfectly, if at all, including health, poverty, and vulnerability.

2.3.4.1 Health Indicators

Biological responses involved in human disease phenomena are among the most important set of parameters for assessing environmental quality, and measures in support of environmental protection are often justified on the basis of their impact on human health (Moghissi 1994).

Health indicators have been used extensively to monitor the health of populations and are usually defined in terms of health outcomes of interest. The majority of health indicators so far developed, however, have no direct reference to the environment; examples include simple measures of life expectancy or cause-specific mortality rates, where no attempt has been made to estimate any portion of these health outcomes attributable to the environment. An Environmental Health Indicator can be seen as a measure that summarizes, in easily understandable and relevant terms, some aspect of the relationship between the environment and health that is amenable to action (Corvalan 1996). It is a summarized measure both of health outcomes and hazard exposures, which represents an underlying causal relationship between an environmental exposure and a health consequence (Pastides 1995). As with all indicators, appropriate EHIs vary according to the problem and the context.

EHIs can be constructed by linking aggregate data (linkage-based), by identifying environmental indicators with a health linkage (exposure-based), or by identifying health indicators with an environmental linkage (outcome-based). There are special complexities in the identification of EHIs since the incidence of many environmentally related diseases cannot be easily traced back to specific environmental exposures (Kjellström 1995). The Driving forces-Pressure-State-Exposure-Effect-Action framework, which has been proposed by the World Health Organization, is a widely accepted conceptual framework to guide the development of EHIs. The Driving Forces component refers to the factors that motivate and push the environmental processes involved (population growth, technological and economic development, policy intervention, and so on). The drivers result in the generation of pressures, normally expressed through human occupation or exploitation of the environment, and may be generated by all sectors of economic activity. In response to these pressures, the state of the environment is often modified, producing hazards. Exposure refers to the intersection between people and the hazards in the environment. These exposures lead to a wide range of health effects, ranging from well-being through morbidity or mortality (Briggs 1999).

EHIs are needed to monitor both trends in the state of the environment and trends in health resulting from exposures to environmental risk factors. They are useful also to compare areas or countries in terms of their environmental health status, to assess the effects of policies and other interventions on environmental health, and to investigate potential links between environment and health (Briggs 1999). EHIs use a variety of units, but many are expressed in disability-adjusted life years: the sum of life years lost due to premature mortality and years lived with disability, adjusted for severity (Murray 1994, 1997).

Usable EHIs depend heavily on the existence of known and definable links between environment and health. Difficulties in establishing these relationships (due, for example, to the complexity of confounding effects and the problems of acquiring reliable exposure data) inhibit the practical use of many potential indicators and make it difficult to establish core indicator sets (Corvalan 2000). Thus, the presence of environmental changes does not translate automatically into health outcomes, and the incidence of many environmentally related diseases cannot be easily traced back to specific environmental exposures. Many broader environmental issues, such as deforestation, loss of biodiversity, soil degradation, and climate change have a much less direct link to health. Although the effects of ecosystem disturbance on human health may be relatively direct, they may also occur at the end of long, complex causal webs, dependent on many intermediate events. When these effects are subtle and indirect, often entailing complex interactions with social conditions, their measurement through indicators is often difficult.

WHO, by assigning weight factors in the form of estimated environmental fraction of reported DALYs for relevant diseases, has estimated that 23% of the global burden of disease is related to environmental factors (WHO 1997).

Sets of specific EHIs have been proposed to monitor both environmental quality and population health levels on a national basis, encompassing different types of hazards (chemical, physical, and biological) and modifications in several ecosystems, such as forests, agroecosystems, and urban ecosystems (Confalonieri 2001). In addition, indicators have recently been proposed to monitor the interactions between human health effects and the quality of specific ecosystems, including oceans (Dewailly 2002), freshwater ecosystems (Morris 2002), and urban systems (Hancock 2002). Table 2.7 shows simple examples of how changes in ecosystem services generate hazards to human health and how these can be measured by EHIs.

Health impact assessment provides a framework and a systematic procedure to estimate the health impact of a proposed intervention or policy action on the health of defined population groups. HIA produces hypothetical health trade-offs of adopting different courses of action (Scott-Samuel et al. 2001). These estimates may be converted in monetary values, to facilitate comparisons with non-health impacts. Applying an HIA typically involves a prospective assessment of a program or intervention before implementation, although it may be carried out concurrently or retrospectively. The HIA gathers opinions and concerns regarding the proposed policy, uses knowledge of health determinants regarding the expected impacts of the proposed policy or intervention, and describes the expected health impacts using both quantitative and qualitative methods, as appropriate.

2.3.4.2 Poverty and Equity

Possibly the most closely watched impacts of ecosystem changes are those that pertain to poverty. Although poverty has historically been defined in strictly economic terms, in recent years a broader understanding of poverty has increasingly been used, in which poverty is understood as encompassing not only deprivation of materially based well-being but also a broader deprivation of opportunities (World Bank 2001). The MA conceptual framework recognizes five linked components of poverty: the necessary material for a good life, health, good social relations, security, and freedom and choice.

Despite the broader understanding of poverty, most poverty indicators still pertain to monetary measures of well-being. Income has been most widely used as a poverty indicator. In recent years, however, many analysts have argued that consumption is a better measure, as it is more closely related to well-being and reflects capacity to meet basic needs through income and access to credit. It also avoids the problem of income flows being erratic at certain times of the year—especially in poor agrarian economies—which can cause reporting errors. Income-based poverty indicators are easier to compare with other variables such as wages. They are also more widely collected, in contrast to consumption data that are seldom collected, thereby limiting the possibility of undertaking comparative analyses.

Monetary-based indicators have the further limitation that they cannot reflect individuals' feeling of well-being and their access to basic services. A household's ability to address risks and threats (and hence, its feeling of well-being) can change dramati-

Table 2.7. Examples of Ecosystem Disruption and Environmental Health Indicators

Ecosystem	Service	Change	Hazard	Human Health Outcome	Indicators
Coastal	waste processing	organic overload	microbes	diarrhea; cholera	incidence
Urban	air quality regulation	air pollution	CO; NO_x; SO_2	asthma	morbidity; body burden of metals
Freshwater	water filtration	depletion	poor hygiene	diarrhea	childhood mortality
Tropical forest	regulation of water and nutrient cycles	deforestation	infections	malaria; arbovirus infections	incidence
Agroecosystem	food production	pesticides	toxic exposure	reproduction problems	fertility rates
Freshwater/marine	provision of fish	overharvesting	depletion of fish resource	reduced consumption of fish protein	protein deficiency

cally even as income and consumption remain stable. Factoring in the effect of vulnerability, analysts estimate that monetary-based indicators can understate poverty and inequality by around 25% (World Bank 2001). In response, efforts have been made to develop non-monetary-based poverty indicators such as outcomes relating to health, nutrition, or education, as well as composite indices of wealth (Wodon and Gacitúa-Marió 2001). These alternative poverty indicators, however, face methodological and data collection issues that make comparisons between countries difficult.

Poverty measures are defined relative to a poverty line (the cutoff separating the poor from the non-poor). Many types of poverty measures exist, but the most commonly used are the headcount index (a measure of poverty incidence, which computes the number of people or share of the population below the poverty line), the poverty gap (a measure of the depth of poverty, which describes how far below the poverty line people are), and the squared poverty gap (a measure of poverty severity, which combines both poverty gap and inequality among the poor). A related set of measures is used to measure inequality, including the Gini coefficient (a measure between 0 and 1, with 0 representing perfect equality and 1 perfect inequality) and the Atkinson index (which incorporates the strength of societal preference for equality).

Most countries determine their own poverty line, making international comparisons of poverty data conceptually and practically difficult. Poverty lines in rich countries are characterized by a higher purchasing power than in poorer nations, making comparisons subject to possible inaccurate interpretation (World Bank 2003). In response, an international poverty line was established in order to measure poverty across countries. The dollar-a-day poverty line (this has been updated to $1.08 a day, in 1993 prices) was chosen. It is converted to local currency units using purchasing power parity exchange rates. However, the non-uniform derivation of the PPP changes the relative value of expenditures between countries and may affect poverty comparisons. The World Bank, for example, uses the PPP-based international poverty line to arrive at comparable aggregate poverty estimates across countries, but it relies mostly on national poverty lines in its poverty analysis.

Reliable and consistent poverty analyses require uniform and high-quality data that are in many cases—especially in developing countries—not available. The Living Standards Measurement Study program was established to develop methods to monitor progress in improving standards of living, in identifying the impacts of policy reforms on well-being, and in establishing a common language by which research proponents and policy-makers could communicate (Grosh and Glewwe 1995). LSMS surveys are used to gather data on a gamut of household activities, many of which are used as poverty indicators. Well-being is measured by consumption; hence in most LSMS research on poverty, measurement of consumption is heavily emphasized in the surveys. With the strong interest in addressing poverty issues in the context of sustainable development, there are current efforts to expand the scope of the LSMS surveys to include variables pertaining to natural resource and environmental management. Exploratory efforts are being undertaken to possibly include a module on environmental health in the LSMS research.

The link between poverty and ecosystem services is established by monitoring changes in ecosystem services and observing how they change poverty measures. The issues of whether the poor are agents or victims of environmental degradation (or both) and of possible trade-offs between ecosystem condition and the well-being of the poor are both burning topics among scholars and policy-makers (Reardon and Vosti 1997; World Bank 2002). Recent work has documented that the poor tend to rely heavily on goods and services provided by the environment and thus are particularly vulnerable to their degradation (Cavendish 1999; Vedeld et al. 2004).

2.3.4.3 Other Indicators

A great number of other indicators can be used to assess various dimensions of human well-being. For example, several indicators help measure progress toward achieving the Millennium Development Goals in addition to the poverty and health indicators just described (World Bank 2002). Adult literacy rates measure educational attainment, and indicators such as net enrollment ratios in primary education or the proportion of students starting grade 1 who reach grade 5 can measure progress toward the goal of universal primary education (MDG 2). The ratio of girls to boys at various levels of education, the ratio of literate females to males, the share of women in nonagricultural employment, and the share of seats in parliament held by women can be used to measure progress toward the goal of promoting gender equality (MDG 3). And maternal mortality ratios and the proportion of births attended by skilled personnel can be used to measure progress toward improving maternal health (MDG 5). These and many other indicators can provide valuable insights, but they are often difficult to relate to ecosystem condition as they are also affected by many other factors. (Note that risk and vulnerability indicators are discussed in Chapter 6.)

2.3.5 Aggregate Indicators of Human Well-being

Several indicators are in use as aggregate indicators of human well-being. The most commonly used, of course, is the gross domestic product, which is a measure of economic activity. This indicator has long been known to be imperfect, even for the narrow purpose of measuring economic activity, let alone as a measure of overall well-being. The limitations of GDP as an indicator have led to substantial efforts to improve it and to develop alternatives.

The linkage between human well-being and national accounting is not particularly straightforward, since GDP, for example, includes both consumption of produced goods—yielding direct benefits for well-being—and investment in physical capital—yielding future benefits for well-being. Moreover, many factors, including the enjoyment of environmental amenities, are not captured in the value of consumption recorded in the national accounts.

Recent results in the theory of environmental accounting make the linkage between asset accounting and well-being explicit. Hamilton and Clemens (1999) show that there is a direct link between the change in the value of all assets (including produced and natural assets) and the present value of social well-being: declining asset values, measured at current shadow prices, imply future declines in social well-being. Dasgupta and Mäler (2000) and Asheim and Weitzman (2001) have extended these results. The World Bank has been publishing estimates of adjusted net saving for roughly 150 countries since 1999 (World Bank 2003). Relying on internationally available data sets, these estimates adjust traditional measures of saving to reflect investments in human capital; depreciation of produced capital; depletion of minerals, energy, and forests; and damages from emissions of carbon dioxide.

Efforts to develop alternative indicators of well-being include composite indices that capture the multidimensionality of well-being. Early attempts to develop composite indices include the Weighted Index of Social Progress (Estes 1984, 1988) and the Physical Quality of Life Index (Morris 1979). More recently, the Human Development Index (UNDP 1998, 2003), which combines measures of life expectancy, literacy, education enrollment, and GDP per capita, has been widely used. The Human Poverty Index is similar, but with different variables for industrial and developing countries, while the Gender-related Development Index adjusts for disparities in achievement for men and women (UNDP 2003). None of these indicators include environmental variables explicitly. One indicator that does is the Calvert-Henderson Quality of Life Indicator, which includes measures of environmental, social, and economic conditions (Flynn 2000; Henderson 2000).

Composite indicators suffer from the arbitrariness of the weighting of their different components, however. Some authors prefer to simply list the components individually, without attempting to aggregate them into a single measure. Thus the World Bank provides a wide selection of indicators in its annual *World Development Indicators* publication (World Bank 2004), and UNDP provide a variety of indicators in addition to the aggregated HDI in the annual *Human Development Report* (UNDP 2003). Many of these indicators have substantial limitations from the perspective of the MA, as they are extremely difficult to relate to environmental conditions.

2.3.6 Intrinsic Value

Economic valuation attempts to measure the utilitarian benefits provided by ecosystems. In addition, many people ascribe ecological, sociocultural, or intrinsic values to the existence of ecosystems and species and, sometimes, to inanimate objects such as "sacred" mountains.

Some natural scientists have articulated a theory of value of ecosystems in reference to the causal relationships between parts of a system—for example, the value of a particular tree species to control erosion or the value of one species to the survival of another species or an entire ecosystem (Farber et al. 2002). At a global scale, different ecosystems and their species play different roles in the maintenance of essential life-support processes (such as energy conversion, biogeochemical cycling, and evolution). The magnitude of this ecological value is expressed through indicators such as species diversity, rarity, ecosystem integrity (health), and resilience. The concept of ecological value is captured largely in the "supporting" aspect of the MA's definition of ecosystem services.

What might be called sociocultural value derives from the value people place on elements in their environment based on different worldviews or conceptions of nature and society that are ethical, religious, cultural, and philosophical. A particular mountain, forest, or watershed may, for example, have been the site of an important event in their past, the home or shrine of a deity, the place of a moment of moral transformation, or the embodiment of national ideals. These values are expressed through, for example, designation of sacred species or places, development of social rules concerning ecosystem use (for instance, "taboos"), and inspirational experiences.

For many people, sociocultural identity is in part constituted by the ecosystems in which they live and on which they depend—these help determine not only how they live, but also who they are. To some extent, this kind of value is captured in the concept of cultural ecosystem services and can be valued using economic valuation techniques. To the extent, however, that ecosystems are tied up with the very identity of a community, the sociocultural value of ecosystems transcends utilitarian preference satisfaction. These values might be elicited by using, for example, techniques of participatory assessment (Campell and Luckert 2002).

The notion that ecosystems have intrinsic value is based on a variety of points of view. Intrinsic value is a basic and general concept that is founded on many and diverse cultural and religious worldviews. Among these are indigenous North and South American, African, and Australian cultural worldviews, as well as the major religious traditions of Europe, the Middle East, and Asia. In the Judeo-Christian-Islamic tradition of religions, human beings are attributed intrinsic value on the basis of having been created in the image of God. Some commentators have argued that plant and animal species, having also been created by God and declared to be "good," also have intrinsic value on the same basis (Barr 1972; Zaidi 1981; Ehrenfeld and Bently 1985).

In some American Indian cultural worldviews, animals, plants, and other aspects of nature are conceived as relatives, born of one universal Mother Earth and Father Sky (Hughes 1983). The essential oneness of all being, *Brahman,* which lies at the core of all natural things, is basic to Hindu religious belief (Deutch 1970). Closely related to this idea is the moral imperative of *ahimsa,* non-injury, extended to all living beings. The concept of *ahimsa* is also central to the Jain environmental ethic (Chapple 1986).

In democratic societies, the modern social domain for the ascription of intrinsic value is the parliament or legislature (Sagoff 1998). In other societies a sovereign power ascribes intrinsic value, although this may less accurately reflect the actual values of citizens than do parliamentary or legislative acts and regulations. The metric for assessing intrinsic value is the severity of the social and legal consequences for harming what society has deemed to be intrinsically valuable.

2.4 Assessing Trade-offs in Ecosystem Services

The challenge to decision-making is to make effective use of new information and tools in this changing context in order to improve the decisions that intend to enhance human well-being and provide for a sustainable flow of ecosystem services. Perhaps the most important traditional challenge in decision-making about ecosystems is the complex trade-off faced when making decisions that will negatively affect or otherwise alter ecosystems. Increasing the flow of one service from a system, such as provision of timber, may decrease the flow from others, such as carbon sequestration or the provision of habitat. In addition, benefits, costs, and risk are not allocated equally to everyone, so any intervention will change the distribution of human well-being—another trade-off. Improved provision of appropriate information can help in assessing the trade-offs among ecosystem services resulting from policy decisions.

Understanding the impact of ecosystem management decisions would be simplest if all impacts were expressed in common units. If information on the impact of ecosystem change is presented solely as a list of consequences in physical terms—so much less provision of clean water, perhaps, and so much more production of crops—then the classic problem of comparing apples and oranges applies.

The purpose of economic valuation is to make the disparate services provided by ecosystems comparable to each other by measuring their relative contribution to human well-being. As utility cannot be measured directly, economic valuation usually attempts to measure all services in monetary terms. This is purely a matter of convenience, in that it uses units that are widely recognized, saves the effort of having to convert values already expressed in monetary terms into some other unit, and facilitates comparison with other activities that also contribute to well-being, such as spending on education or health. In particular, it puts the impacts of ecosystem change into units that are readily understood by decision-makers and the lay public. When all impacts of ecosystem change are expressed in these terms, then they can readily be introduced into frameworks such as cost-benefit analysis in order to assess policy alternatives.

Other metrics are occasionally proposed. Some analysts, for example, have advocated the use of energy units (Odum and Odum 1981; Hall et al. 1986), arguing that as all goods and services are ultimately derived from natural resources by expending energy, energy is the real source of material wealth. These approaches can provide valuable insights into particular issues. For purposes such as the MA, however, these approaches have several disadvantages—in particular, they have no direct link to human well-being, and they require a considerable effort to convert a wide variety of impacts into common units.

Efforts to place everything into common units will necessarily remain incomplete, however, sometimes because of lack of data and sometimes because value arises not from utilitarian benefits but from intrinsic value or from another source of value. Societies have many objectives, only some of them purely utilitarian. Furthermore, the value of an ecosystem service varies, depending on whether a critical threshold for ecosystem condition or human well-being is crossed (Farber et al. 2002). In other words, placing everything into common units is sometimes impossible and frequently undesirable. It is important to stress, however, that even incomplete efforts to express impacts in common units can be helpful by reducing the number of different dimensions that need to be taken into considerations.

Graphical depictions of the trade-offs in ecosystem services associated with alternative policy options can provide useful input to decision-makers. ''Spider diagrams'' such as that in Figure 2.4 can depict the amount of ecosystem services associated with different management alternatives. For example, Figure 2.4 depicts hypothetical trade-offs among five ecosystem services associated with an expansion of cropland in a forested area: food production, carbon sequestration, species richness, soil nutrients, and base streamflow. Comparison of the ecosystem services available before forest conversion to cropland with the services after forest conversion allows a decision-maker to account for the full suite of ecosystem services affected by the conversion. The approach requires quantifiable and measurable indicators for each of the services depicted. The quantities depicted can be an absolute measure (such as tons of carbon stored) relative to a previous quantity, to a relevant average quantity (for the area, for instance, or for the biome), or to an ideal ''sustainable'' amount.

The degree to which the diagram effectively communicates trade-offs in ecosystem services depends on the explicit definition of the values on the axes and the ability to quantify them. A series of diagrams for varying time since forest clearing and for varying

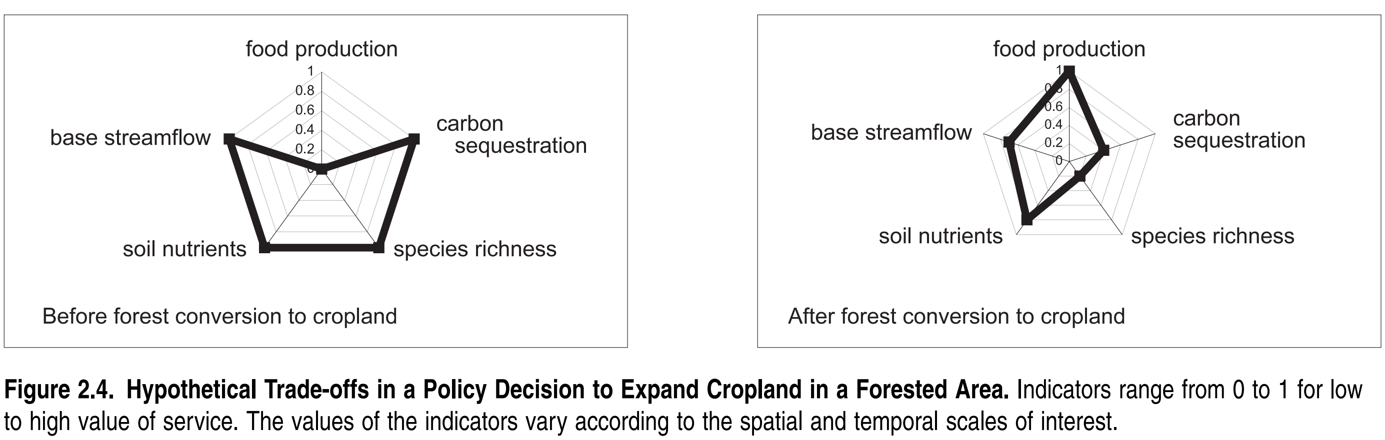

Figure 2.4. Hypothetical Trade-offs in a Policy Decision to Expand Cropland in a Forested Area. Indicators range from 0 to 1 for low to high value of service. The values of the indicators vary according to the spatial and temporal scales of interest.

spatial scales of interest could be used to inform decision-makers about the effects on ecosystem services for the varying scales of analysis. When a large number of management alternatives are to be compared, they can be portrayed either in a series of spider diagrams or across all management alternatives, as in Figure 2.5 (Heal et al. 2001a).

Depictions of ecosystem services associated with predefined management alternatives, as in Figures 2.4 and 2.5, are simple and readily communicable to decision-makers but are often unable to account for non-linearities and thresholds in responses of ecosystem services to management decisions. When such phenomena are present, figures such as Figure 2.6 can help assess choices. For example, application of nitrogen fertilizer involves a trade-off between increasing crop yields and decreasing coastal fisheries if nitrate leaching leads to hypoxia in downstream coastal locations, as it has in the Mississippi Delta (Donner and Kucharik 2003). Balancing an objective of maximum crop yields with minimum damage to coastal fisheries requires knowledge of the response curves of each service to nitrogen fertilizer application. In this example, fertilizer application beyond point "A" results in negligible increase in crop yield but substantial nitrate leaching. A decision to apply fertilizer greater than point "A" trades small increases in crop yield for large increases in nitrate leaching. A decision to apply fertilizer less than point "A" trades small decreases in nitrate leaching for forgone large increases in crop yield. To the extent that the shape of the response curves can be quantified, management alternatives can account for these types of nonlinear responses to determine the most desirable alternative.

Portraying interactions among multiple ecosystem services graphically quickly becomes complex and unwieldy. Heal et al. (2001a) suggest constructing "production possibility frontiers" to model combinations in the amounts of ecosystem services possible to achieve a management objective. For example, possible combinations of ecosystem services such as carbon storage and timber production can be modeled to achieve varying levels of water purification. The optimal mix of these services can then be selected, depending on the management objectives.

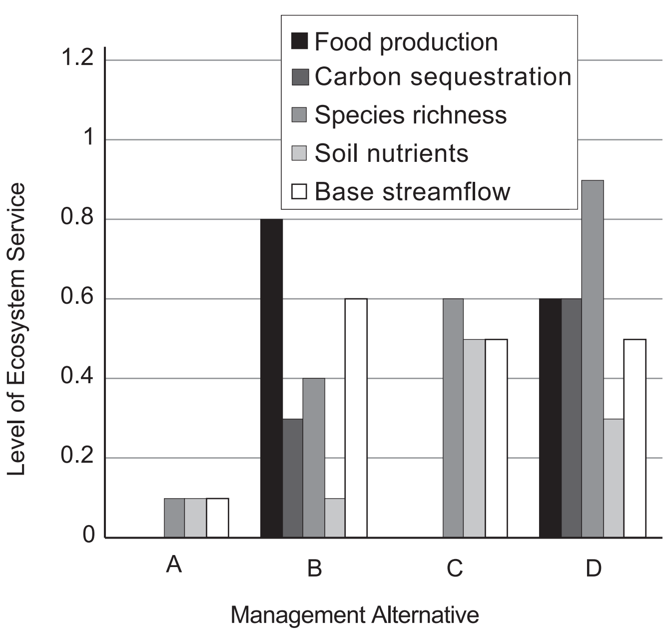

Figure 2.5. Portrayal of Hypothetical Trade-offs in Ecosystem Services Associated with Management Alternatives for Expanding Cropland in a Forested Area. Indicators range from 0 to 1 for low to high value of service. See text for management alternatives. (Adapted from Heal et al. 2001b).

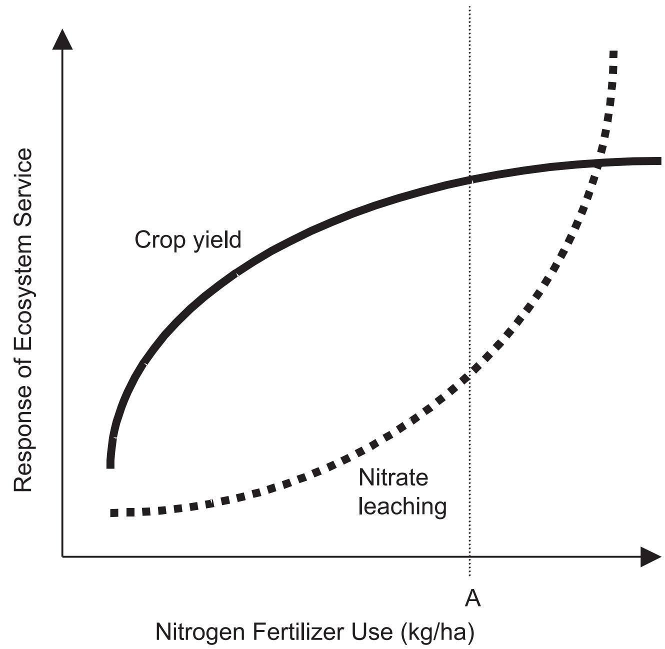

Figure 2.6. Example of Nonlinear Responses of Two Ecosystem Services (Crop Yields and Coastal Fisheries) to Application of Nitrogen Fertilizer

Multicriteria analysis provides another formal framework to help assess choices in the presence of multiple, perhaps contradictory, objectives (Falconí 2003). In a multicriteria analysis, a matrix is constructed showing how each of the alternatives under consideration ranks relative to the other alternatives, according to each criterion. This impact matrix, which may include quantitative, qualitative, or both types of information, allows the best alternative to the decision or analysis problem to be found (Munda 1995; Martínez-Alier et al. 1998). A vast number of multicriteria methods have been developed and applied for different policy purposes in different contexts (Munda 1995). The main advantage of such models is that they make it possible to consider a large number of data, relations, and objectives that are generally present in a specific real-world decision problem, so that the decision problem at hand can be studied in a multidimensional fashion. When different conflicting evaluations are taken into consideration, however, a multicriteria problem is mathematically ill defined. The application of the different methods can lead to different solutions. In some cases, solutions that satisfy multiple objectives may not be possible.

Consideration of the trade-offs involves clear definitions about the spatial and temporal scales of interest. How are future impacts on ecosystem services included in the analysis? Over what time frame should these impacts be considered? Does the alteration in ecosystem services affect human well-being distant in space from the ecosystem change (such as through downstream effects or atmospheric transport)? How are impacts that cross administrative or ecosystem boundaries incorporated in the analysis? Assessments need to be conducted within a scale domain appropriate to the processes or phenomena being examined. Cost-benefit analysis has often fallen short in the past in part because the spatial and temporal boundaries it used did not encompass all the impacts of the proposed interventions (Dixon et al. 1994). This same weakness applies to all assessment methodologies: they will only be meaningful if the spatial and temporal scales of the analysis have been carefully defined. Too narrow a definition of either could result in a misperception of the problems. For example, if soil nutrients decline over time under agricultural use, the perceived impact on that dimension depends crucially on the time period chosen for the indicators.

Appendix 2.1. Core Data Sets Used by the MA to Assess Ecosystem Condition and Trends

The Millennium Ecosystem Assessment has involved the development and distribution of a range of data sets and indicators. Although the overall MA products primarily consist of syntheses of findings from existing literature, the data and indicators developed or presented within the MA play important roles both in presenting information on the links between ecosystems and human well-being and in establishing year 2000 ''baseline'' conditions for reference in future global and sub-global assessments.

For many central themes of the MA, there are multiple available data sets on which elements of the assessment could be based and from which different conclusions could be drawn. For example, there is a range of land cover data sets available based on information from different satellite sensors and interpretation techniques, from which different statistics on land cover could be generated. To ensure consistency of analysis and comparability of results across the chapters and working groups of the MA, a small number of MA ''core data sets'' were selected. (See Appendix Table 2.1.) Although chapter teams also made use of alternative data sets, applicable findings are in each case also presented based on an analysis with the various core data sets, and the strengths and weaknesses of these data sets are assessed for the particular application in the chapters.

A description of the choice of MA systems, the main reporting unit for the Condition and Trends Working Group, can be found in the Preface. Appendix Table 2.2 presents the updated system boundary definitions, adding detail to the brief system descriptions given in Box 1–3 of Chapter 1.

Data management procedures were developed for the use of data sets in the MA. A Web-based data catalogue recorded metadata for all data sets used in the MA. Data Archives were established at CIESEN, the World Data Center For Biodiversity and Ecology, and UNEP–WCMC for all data in categories 4–6 of Appendix Table 2.3, as well as for some data in category 2 if they were used for a significant portion of analysis in a particular chapter. MA archived data are freely accessible to any user, and all archived data sets are accompanied by metadata in the ISO metadata standard (ISO 19115: Geographic Information).

Appendix Table 2.1. Summary of MA Core Datasets

Core Dataset	Brief Description	Lead Agencies
Global land cover	Global Land Cover 2000 dataset; a global product of land cover in year 2000, based on SPOT Vegetation satellite data	EU JRC, with regional networks
Human population density	an updated Gridded Population of the World dataset, referenced to year 2000, and including a rural/urban split, including a point database of human settlements >5,000 people, an urban mask (polygons), and a complete urban-rural gridded surface	CIESIN, with World Bank and IFPRI
Protected areas	the 13th UN List of Protected Areas, from which a "snapshot" of the extent of Protected Areas in the year 2000 has been generated, as a baseline dataset for the MA	UNEP-WCMC, with WCPA
Subnational agricultural statistics	sub-national time series and single year crop production data including area, production, and yield, available for the globe	IFPRI, with wider consortium
Climate	0.5-degree dataset of monthly surface climate extending from 1901 to 2000 over global land areas, excluding Antarctica 10-minute mean monthly surface climate grids for the 1961–90 period covering a similar area	University of East Anglia CRU and University of Oxford, UK
Human well-being indicators	sub-national infant mortality, malnutrition, and GDP data; global data, although malnutrition index only available for the developing world	CIESIN
Areas of rapid land cover change	a synthesis of the knowledge of areas affected by rapid land cover change during the last 20 years for various change classes, including deforestation, cropland and pasture expansion, soil degradation and desertification, urban expansion, and exceptional fire events	IGBP/IHDP, LUCC, GOFC/GOLD
Global MA reporting "units"	datasets delineating MA system boundaries (see Appendix Table 2.2), biomes and biogeographical realms, and socioeconomic regional reporting units	various

Appendix Table 2.2. MA System Boundary Definitions

MA System	Description
Coastal	The area between the interpolated 50 m bathymetry and 50 m elevation contours from the ETOPO2 dataset. The 50 m inland contour is constrained to a maximum distance of 100 km.
Cultivated	Agricultural classes from version 2 of the Global Land Cover Characteristics Dataset. Cropland, pasture, and mosaic (or mixed) agriculture and other land use classes are included.
Dryland	A subset of the aridity zone map published in the World Atlas of Desertification. Aridity zones are derived from an Aridity Index calculated as the ratio of precipitation to potential evapotranspiration. The zones hyper-arid, arid, semiarid, and dry subhumid are included in the dryland system.
Forest and woodland	Derived from the Global Land Cover 2000 Dataset. Extracted classes are broadleaved, needle-leaved, mixed tree cover, regularly flooded (such as mangroves) and burnt tree cover, and a mosaic tree cover/other natural vegetation class (classes 1 to 10 of the global classification).
Inland water	Includes major rivers, wetlands, lakes, and reservoirs as compiled in the Global Lakes and Wetlands Database–Level 3.
Island	Oceanic and coastal islands as defined by ESRI's ArcWorld Country Boundaries dataset. Approximately 11,925 islands are represented and include those listed as members of the Alliance of Small Island States and the Small Island Developing States Network.
Marine	The marine system boundary is defined from the interpolated 50 m bathymetry (from the ETOPO2 dataset) seaward. Longhurst's biome classification provides subsystem categorizations.
Mountain	Derived from UNEP-WCMC's mountain dataset, using criteria of altitude, slope, and local elevation range. Altitudinal life zones form subsystem reporting units.
Polar	Arctic and sub-arctic vegetation types define the northern hemisphere portion of the polar system. Vegetation types are delineated from a combination of global and regional land cover maps from remote imagery. Antarctica forms the southern portion of the polar system.
Urban	Derived from the Global Land Cover 2000 Dataset artificial surfaces class (class 22 in the global legend).

Appendix Table 2.3. Data Handling Procedures in the MA

Data Application in the MA	Data Handling Procedures
1. Peer-reviewed or validated datasets cited in MA reports	full citation in MA report
2. Peer-reviewed or validated datasets used in MA analysis (e.g., to calculate area, quantity), map, or table but unmodified	full citation in MA report included in MA Data Catalog may be included in datasets available for online access as part of MA outreach
3. Non-peer-reviewed datasets cited in MA reports	dataset critically assessed; quality and validity of the dataset reviewed by chapter team before incorporating results from the source into an MA Report following materials sent to the Working Group Technical Support Unit: title of dataset; location (URL if available); institution responsible for maintaining the data; information on the availability of the data to other researchers; contact details for one or two people who can be contacted for further information about the source
4. Non-peer-reviewed datasets used in MA analysis, map, or table but unmodified	procedures in category 3 followed included in MA Data Catalog included in MA Data Archive if possible (particularly if a key dataset for the analysis) may be included in datasets available for online access as part of MA outreach
5. Data modified in an MA analysis or new datasets produced through existing peer-reviewed data; considered an "MA Dataset"	dataset critically assessed; quality and validity of the dataset reviewed by chapter team before incorporating results from the source into an MA Report. MA Metadata Standards followed included in MA Data Catalog and MA Data Archive made freely available to other users
6. MA Core Datasets	MA Metadata Standards followed included in MA Data Catalog and Data Archive made freely available to other users
7. MA Heritage Datasets — datasets representing a valuable "baseline" condition for year 2000 (e.g., NDVI data)	MA Metadata Standards followed included in MA Data Catalog and MA Data Archive made freely available to other users

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