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Measuring long-term ecological changes in densely populated landscapes using current and historical high resolution imagery

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Abstract

Long-term ecological changes within densely populated landscapes account for a growing share of global environmental change. Measuring the causes and consequences of these changes remains a challenge because of their fine spatial scale and complexity. Here, we measure long-term ecological changes, circa 1950 to 2002, within six 1 km² sites in densely populated rural China and in urban and suburban Baltimore, Maryland, USA using a standardized procedure for fine-scale feature-based ecological mapping from high spatial resolution (<1 m) imagery. The median size of ecologically distinct landscape features (ecotopes) mapped by this procedure was just 520 m², though size, count and perimeter of features varied considerably both within and between sites. Land management and vegetation cover changed substantially, over 28% to 87% of site areas, but most of this change occurred in small patches with area <4000 m². Landscape complexity also increased over time by the fragmentation of landscapes into a larger number of smaller features with an increasing diversity of ecotope classes. Detailed analysis of fine-scale landscape transformations helped identify the causes and consequences of ecologically significant changes within and across sites, including unexpected increases in perennial vegetation cover and the linkage of impervious surface area with population density. These and other results demonstrate the general utility of anthropogenic ecotope mapping as a tool for cross-site comparison and sampled regional estimates of long-term ecological changes within densely populated landscapes.

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1. Introduction

Densely populated rural, urban and suburban landscapes now cover as great an extent of Earth’s land surface as do tropical rainforests and many other globally important ecosystems (Achard et al., 2002; Ellis, 2004; Foley et al., 2003). Ecosystem processes and their spatial patterns within these anthropogenic landscapes are profoundly altered by human activity and contribute disproportionately more per unit area to global changes in climate, biogeochemical cycles and biodiversity (Foley et al., 2005; Hope et al., 2003; Kalnay & Cai, 2003; Kaye et al., 2004; Matson et al., 1997; Vitousek et al., 1997). Given their global impacts and the fact that most humans live within them (~5.8 of 6.2 billion persons live in areas with ≥25 persons km⁻²; Oak Ridge National Laboratory, 2004), the measurement and mediation of long-term ecological changes within densely populated landscapes are a matter of serious global, regional, and local concern (Foley et al., 2005; Vitousek et al., 1997).
In comparison with deforestation, urban expansion, and other extensive landscape transformations that often precede dense human occupation, the causes and consequences of long-term ecological changes within densely populated landscapes are only beginning to be understood (Foster, 1992; Green et al., 2005; Grimm et al., 2000; Heilig, 1994; Hope et al., 2003; Kaye et al., 2004; Lambin et al., 2001; Turner et al., 1994). This is not surprising, considering that landscape transformations within densely populated landscapes incorporate a wide variety of complex land management practices that are characterized by fine-scale changes in landscape structure (<30 m) caused by the creation, transformation, and abandonment of anthropogenic features with distinct boundaries, such as buildings, roads, yards and small agricultural plots (Ellis et al., 2000a; Foster, 1992; Jensen & Cowen, 1999). These complex fine-scale changes in land use are a challenge to measure by conventional remote sensing approaches (Cihlar & Jansen, 2001; Forster, 1985; Guindon et al., 2004; Lobell & Asner, 2004; Price, 2003; Rindfuss et al., 2004; Thomas et al., 2003; Woodcock & Strahler, 1987) and are usually left out of global and regional ecological change estimates, potentially introducing substantial errors into these (Easterling, 1997; Houghton, 2003; House et al., 2003; Hutt et al., 2003; Johnes & Butterfield, 2002; Turner et al., 1994).

Long-term ecological changes within anthropogenic landscapes are the combined result of changes in landscape structure, land management practices, and ecosystem processes (Binford et al., 2004; Ellis, 2004; Grimm et al., 2000; Kaye et al., 2004; Matson et al., 1998; Rindfuss et al., 2004; Turner et al., 1994). Usually, these are measured by different methods, each with different land units, and the results are then integrated to estimate ecological change (review by Rindfuss et al., 2004). However, the simplest way to link these measurements is to use the same land units when measuring changes in landscape structure, when obtaining land management data by interviewing local land managers, and when sampling and measuring ecological parameters in the field (Ellis, 2004). To accomplish this, landscapes must first be stratified into ecologically distinct features identifiable to both land managers and ecologists in the field and in imagery, a task best accomplished by field-validated interpretation of ecologically distinct features in high spatial resolution imagery (<1 m; Ellis, 2004; Jensen & Cowen, 1999; Thomas et al., 2003). Moreover, to measure ecological changes over the long term (>50 y) and across the full range of anthropogenic landscapes, from rural to urban, from floodplain to mountainous, and from pre-industrial to contemporary, a standardized a priori ecological classification procedure is necessary that can produce consistent results from historical aerial photographs and other sources of high spatial resolution imagery, including IKONOS and Quickbird (Cihlar & Jansen, 2001; Jansen & Gregorio, 2002; Kadmon & Harari-Kremer, 1999; Sawaya et al., 2003; Thomlinson et al., 1996).

This study applies the first standardized fine-scale ecological mapping procedure designed explicitly for densely populated landscapes by measuring long-term ecological changes, circa 1950 to 2002, within six densely populated ecological research sites across rural China and in urban and suburban Baltimore, Maryland, USA. Consistent with the ecosystem concept, which combines biotic and abiotic components within a single unit (the ecosystem), our procedure stratifies landscapes into ecologically distinct units (ecotopes) based on a combination of biotic and abiotic classification terms (Ellis et al., 2000a; Klijn & Udo De Haes, 1994). The procedure maps ecotope features based on relatively stable boundaries between ecologically distinct classes of land management and vegetation cover observable in the field at ground level, facilitating ecological fieldwork and land management interviews and providing a consistent mapping product from the complex layered mixture of land cover and land use that predominates in densely populated landscapes (Fig. 1; Sawaya et al., 2003; Jensen & Cowen, 1999). To our knowledge, all existing a priori standards for thematic environmental mapping are either lower resolution pixel based methods (30–1000 m; e.g. Cruickshank & Tomlinson, 1996; Hansen et al., 2000; Homer et al., 2004; Latifovic et al., 2004) or are site- or sensor-based systems that were not designed for consistent long-term change measurements across different regions of the world prior to the 1990s (e.g. Akbari et al., 2003; Anderson et al., 1976; Freeman & Buck, 2003; Kadmon & Harari-Kremer, 1999; Lu et al., 2004; Thomas et al., 2003; Thomlinson et al., 1999). Moreover, we know of no other existing ecological mapping system designed specifically to stratify anthropogenic landscapes into ecologically distinct features that are as useful for surveying local land management practices as they are for sampled ecological measurements in the field.

There are several key challenges in measuring fine-scale ecological changes based on field-validated feature-based mapping. First of all, like land management surveys and ecological sampling in the field, fine-scale mapping is highly resource-intensive. As a result, regional measurements of fine-scale ecological changes are best accomplished by the parsimonious application of these methods within regionally stratified sampling designs linked to data obtained by coarser-resolution remote sensing, such as area frame sampling (Achard et al., 2002; Cihlar et al., 2000; Ellis, 2004; Gallego, 2004; Gallego et al., 1994; Hutt et al., 2003; Price, 2003;...
Our fine-scale mapping procedure was therefore designed expressly for application within sample cells selected by regional analysis, so that fine-scale measurements within sample cells, such as crop area, tree biomass, and fertilizer inputs can be linked directly with regional remote sensing data to estimate the causes and consequences of fine-scale ecological changes at regional scales (Ellis, 2004).

Another issue in measuring fine-scale ecological changes is that changes measured by comparing current and historical maps are highly sensitive to “false change” errors caused by misregistration between maps (Foody, 2002; Townshend et al., 1992). This error can be avoided by estimating changes in ecological map class areas across entire sample cells, if sample cells are large enough (Wang & Ellis, 2005b). Another major source of error is disagreement between trained interpreters in both feature mapping (shape-based error) and feature classification, even when maps are field-validated by interpreters (Cherrill & McClean, 1999; Ellis & Wang, submitted for publication; Green & Hartley, 2000; Powell et al., 2004). Though interpreter error is unavoidable, it can be reduced by standardized collaborative training to calibrate results across interpreters, by scale-explicit rules for mapping and classification, and by the continuous centralized supervision of local mapping efforts (Cherrill & McClean, 1999; Cherrill et al., 1995; Ellis & Wang, submitted for publication; Powell et al., 2004). Most importantly, interpreter error in map class area estimates can be quantified to predict conservative error intervals that prevent false detection of error in map class area estimates can be quantified to predict ecological changes at regional scales.

This study will demonstrate that fine-scale change measurements reveal substantial and often unexpected long-term ecological changes that would not be observable by coarser scale approaches. The general utility of a priori hierarchical ecological classification will be established by comparing land use and land cover changes in densely populated landscapes across China and in the USA, and by comparing long-term changes in land use and land cover across sites. The causes and consequences of pronounced long-term changes in land use and land cover across sites are then investigated using the more detailed information available in fine-scale ecotope maps of densely populated landscapes.

2. Methods

2.1. Sites and imagery

Six square 1 km² sites were selected for study across a broad range of environmental, developmental, and population density conditions within existing ecological research sites in the USA and China (Table 1). Prior to site selection, regions were stratified into 500 × 500 m cells by imposing a 500 m² sampling frame across the landscape. This provided sample units practical both for fieldwork and integration with regional and global remote sensing data (Ambrosio Flores & Iglesias Martinez, 2000; Ellis, 2004; Gallego et al., 1994; Townshend & Justice, 1988). 1 km² sites within four environmentally distinct regions of China (Gaoyi, Jintang, Yiyang, and Dienbai) were assembled from four adjacent 500 m² sample cells as part of a regional sampling design in a study of long-term biogeochemical changes across densely populated rural China (Ellis, 2004). US 1 km² sites were assembled from adjacent 500 m² cells gridded across an urban watershed research site in Baltimore City (Baltimore) and across the footprint of a carbon flux tower in suburban Baltimore County (Cub Hill) as part of the Baltimore Ecosystem Study (http://www.beslter.org/).

IKONOS 4 band pan-sharpened 1 m resolution GEO imagery (http://www.spacemaging.com) was acquired across China sites in the winter of 2001/2002 and orthorectified from ground control points obtained by submeter accuracy Global Positioning Systems (GPS) as described by Wang and Ellis (2005a). Historical aerial photographs for China were obtained from the U.S. National Archives and Records Administration (NARA; RG-373, http://www.archives.gov) and orthorectified with global data (Willmott and Matsuura, 2001).
Table 2
Land FORM classes

<table>
<thead>
<tr>
<th>Code</th>
<th>Name</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>AN</td>
<td>Anthropogenic</td>
<td>Anthropogenic surfaces and structures</td>
</tr>
<tr>
<td>MO</td>
<td>Mounded</td>
<td>Artificial mounds, earthworks and dams &gt;2 m high and 30 m across</td>
</tr>
<tr>
<td>EX</td>
<td>Excavated</td>
<td>Excavations, mines and pits &gt;2 m deep and 30 m across</td>
</tr>
<tr>
<td>TS</td>
<td>Terraced Slope</td>
<td>Artificial terraces on &gt;5% slope</td>
</tr>
<tr>
<td>FP</td>
<td>Floodplain</td>
<td>Alluvial floodplain, &lt;3% slope</td>
</tr>
<tr>
<td>FS</td>
<td>Foot Slope</td>
<td>Level areas at bottom of hills with non-alluvial soils, &lt;5% slope</td>
</tr>
<tr>
<td>BP</td>
<td>Bench Plateau</td>
<td>Natural bench plateau, &lt;15% slope between &gt;30% slopes</td>
</tr>
<tr>
<td>SU</td>
<td>Summit</td>
<td>Hilltop plateau, &lt;15% slope surrounded by &gt;30% slopes</td>
</tr>
<tr>
<td>SL</td>
<td>Sloping</td>
<td>Sloping hillsides, 3–30% slope</td>
</tr>
<tr>
<td>SS</td>
<td>Steep Slope</td>
<td>Steep slopes, &gt;30% slope</td>
</tr>
<tr>
<td>RP</td>
<td>Retention Pond</td>
<td>Artificial runoff retention ponds, &gt;1 m deep</td>
</tr>
<tr>
<td>MA</td>
<td>Marsh</td>
<td>Lentic wetland</td>
</tr>
<tr>
<td>FM</td>
<td>Flowing Marsh</td>
<td>Slowly flowing wetland in channel, usually in floodplain</td>
</tr>
<tr>
<td>MS</td>
<td>Shallow Stream</td>
<td>Rapidly flowing shallow natural watercourse, &lt;30 m wide</td>
</tr>
<tr>
<td>SM</td>
<td>Seasonal Stream</td>
<td>Seasonally flowing natural watercourse, &lt;30 m wide</td>
</tr>
<tr>
<td>SR</td>
<td>Seasonal River</td>
<td>Seasonally flowing natural watercourse, &gt;30 m wide</td>
</tr>
<tr>
<td>WM</td>
<td>Water Margin</td>
<td>Margin of lentic water body</td>
</tr>
<tr>
<td>CM</td>
<td>Canal Margin</td>
<td>Margin of flowing anthropogenic watercourse</td>
</tr>
<tr>
<td>RM</td>
<td>River Margin</td>
<td>Margin of flowing natural watercourse</td>
</tr>
<tr>
<td>PA</td>
<td>Small Pond</td>
<td>Lentic water body, &lt;30 m wide</td>
</tr>
<tr>
<td>PB</td>
<td>Large Pond</td>
<td>Anthropogenic lentic water body, &gt;30 m wide</td>
</tr>
<tr>
<td>RE</td>
<td>Reservoir</td>
<td>Dammed water body, &gt;30 m wide</td>
</tr>
<tr>
<td>LA</td>
<td>Lake</td>
<td>Natural lentic water body, &gt;30 m wide</td>
</tr>
<tr>
<td>CA</td>
<td>Small Canal</td>
<td>Flowing anthropogenic watercourse, &lt;30 m wide</td>
</tr>
<tr>
<td>CB</td>
<td>Large Canal</td>
<td>Flowing anthropogenic watercourse, &gt;30 m wide</td>
</tr>
<tr>
<td>RS</td>
<td>Stream</td>
<td>Flowing natural watercourse, &lt;30 m wide</td>
</tr>
<tr>
<td>RV</td>
<td>River</td>
<td>Flowing natural watercourse, &gt;30 m wide</td>
</tr>
</tbody>
</table>

2.2. Ecotope mapping and classification

Two scale-explicit standards were developed for ecotope mapping. The Level 1 procedure employed here was designed for relatively rapid current and historical mapping across 500 × 500 m² sample cells, with a single trained interpreter capable of mapping >1 km² in <30 d, including all fieldwork and data processing. An even finer-scale Level 2 procedure was also developed, but was limited to current mapping within Level 1 map samples, the subject of future work (Ellis, 2004). Level 1 ecotope feature polygons were mapped by trained interpreters across 500 m sample cells extended by a 50 m buffer (to minimize edge effects) by three cycles of scale-explicit, rule-based interpretation of imagery by “heads-up” digitizing in a GIS coupled with validation and correction of all features in the field using 1:1200 scale image and ecotope feature maps supported by local land managers and GPS. Feature mapping followed a scale-explicit procedure based on the precision by which different types of anthropogenic ecotope features were identifiable in 1 m resolution imagery and in the field (Cherrill & McLean, 1999; Jensen & Cowen, 1999). First, linear features were mapped (≥2 m width and ≥25 m² area, length ≥4 × width; examples are roads and ditches), followed by hard areal features (≥5 m width and ≥25 m² area with clear edges and homogenous interiors; examples are buildings and water bodies), and lastly the remaining soft features were mapped (≥5 m width and ≥100 m² area with relatively fuzzy edges and variable cores; examples are crop plots and vegetation patches). All ecotope features were corrected by field validation to conform to stable (potentially...
**Table 4**

<table>
<thead>
<tr>
<th>Code</th>
<th>Name</th>
<th>Description</th>
<th>Surface</th>
<th>Herbaceous*</th>
<th>Woody</th>
</tr>
</thead>
<tbody>
<tr>
<td>S</td>
<td>Sealed</td>
<td>Artificial structures and surfaces</td>
<td>&gt;75%</td>
<td>≈&lt; 25%</td>
<td>Variable</td>
</tr>
<tr>
<td>X</td>
<td>Barren</td>
<td>Minerals, permanent snow and ice</td>
<td>&lt;10%</td>
<td>≈&lt; 10%</td>
<td>≈&lt; 10%</td>
</tr>
<tr>
<td>E</td>
<td>Bare soil</td>
<td>Bare soil</td>
<td>&gt;75%</td>
<td>≈&lt; 10%</td>
<td>≈&lt; 10%</td>
</tr>
<tr>
<td>V</td>
<td>Variable</td>
<td>Too variable across years to classify (seasonal</td>
<td>Varies</td>
<td>Varies</td>
<td>Varies</td>
</tr>
<tr>
<td>W</td>
<td>Water</td>
<td>Water surface</td>
<td>&gt;90%</td>
<td>≈&lt; 10%</td>
<td>≈&lt; 10%</td>
</tr>
<tr>
<td>A</td>
<td>Annual</td>
<td>Herbaceous vegetation</td>
<td>&gt;25%</td>
<td>Varies</td>
<td>&lt;60%</td>
</tr>
<tr>
<td>M</td>
<td>Mixed</td>
<td>Mix of herbaceous, open woody and tree cover</td>
<td>&quot;&quot;</td>
<td>&quot;&quot;</td>
<td>&gt;60%</td>
</tr>
<tr>
<td>P</td>
<td>Perennial</td>
<td>Cover by trees, shrubs or other woody perennials</td>
<td>&quot;&quot;</td>
<td>&quot;&quot;</td>
<td>&gt;60%</td>
</tr>
</tbody>
</table>

* Canopy cover (herbaceous, woody) at annual maximum extent, including only cover by plants with main stems rooted within the classified feature, excluding containerized, enclosed, and non-rooted vegetation.

Observable for ≥2 y) land management boundaries at ground level in the field by the interpreter and local land managers in cases where vegetation cover, shadow, or off-nadir imagery confused land-use boundaries in imagery (Fig. 1). Where management or other processes did not produce clear boundaries within areas >1 ha, contiguous vegetation patches larger than 30 m in all dimensions (area >900 m²) were mapped as separate features only when their COVER class differed from surrounding vegetation. This minimum mapping rule was used to limit inconsistency between interpreters in identifying vegetation features with poorly defined edges (Cherrill & McClean, 1999).

After feature mapping, features were classified using a four level a priori ecological classification hierarchy, FORM → USE → COVER → GROUP + TYPE, combining simple landform (FORM; Table 2), land-use (USE; Table 3) and land-cover classes (COVER; Table 4) with a set of more detailed GROUPs of management and vegetation classes stratified into TYPEs. Land FORM classes are based on three hydrologic categories (terrestrial, wetland, and aquatic) that were further stratified by geomorphic shape and edaphic process (Table 2). USE classes define ecologically distinct land management syndromes impacting soils, hydrology, vegetation, livestock, and material input/output (Table 3), while COVER classes describe feature surface cover at ground level (Table 4). GROUP classes comprise 46 land management and vegetation categories that were further divided into TYPEs by a hierarchical system specific to each GROUP, creating ecologically distinct and detailed GROUP + TYPE classes identifiable to both ecologists and land managers in the field. Ecotope classes are defined by combining all four classification levels within each feature. For example, a closed canopy forest (COVER = P = Perennial) of regrowth deciduous trees (GROUP + TYPE = dt02) managed for harvest (USE = T = Forestry) on a gentle slope (FORM = SL = Sloping) is classified as the ecotope “SLTPdt02” (FORM + USE + COVER + GROUP + TYPE).

To calibrate results across sites, interpreters were trained prior to mapping by the repeated blind comparison of their maps against standardized reference maps at 2 different sites. Trained interpreters prepared current ecotope maps for each sample cell by a standard sequence including cell reclassification to catalog local ecotopes and clarify confusing areas, the preparation of an initial map corrected by a field visit, the completion of a fully field-validated map corrected by an additional field visit, and a final field validation visit together with another trained interpreter. All interpreters were centrally supervised for conformance with mapping standards and were in regular communication to resolve mapping issues as they arose. Historical maps were prepared after current maps, and were field-validated in China aided by two local elders per sample cell, by a combination of interviews assisted by 1:1200 scale image maps and field visits together with elders to all confusing areas. Elders selected for historical field validation were aged ≥16 at time of image acquisition (1944/45; age at interview ≥75 y) and had a lifelong history of land management within 500 m of the cell. Cub Hill historical map validation involved a single elder and Baltimore historical map validation was assisted by 1951 Sanborn maps (Sanborn Map Company, 1984). Current mapping of US sites utilized existing planimetric building and road features. Sites in China and Cub Hill were mapped to represent features at time of image acquisition (the standard method); Baltimore maps were corrected to features observed during field validation in 2003. Completed maps were clipped to sample cell boundaries, assembled into 1 km² site maps and checked against a classification database to correct invalid ecotope code combinations. Adjacent features with the same ecotope class were combined, and feature topology was checked and corrected to ensure continuous classified polygon feature layers with an overall map area error tolerance of ±0.05% of sample cell area.

2.3. Error, data quality and change

Overall accuracy of current and historical Level 1 ecotope mapping was >85% across interpreters under the most challenging mapping conditions across sites and time periods (current κ >85%, historical κ >75%), meeting common thematic map accuracy standards (Ellis & Wang, submitted for publication). Changes in map class areas across 1 km² sites were used for long-term change detection, to avoid potential false change error caused by misregistration between maps.
(Wang & Ellis, 2005b). A conservative error model based on larger, more conservative error intervals was used to avoid the false detection of changes and differences in map class areas (Type I error), allowing that smaller changes might go undetected (Type II error; Ellis & Wang, submitted for publication; Schenker & Gentleman, 2001). This model estimated errors from mapped feature perimeter and proved reliable in avoiding false change detection, but introduced more error than normally encountered under less challenging mapping conditions (Ellis & Wang, submitted for publication).

A Monte Carlo (MC) observational uncertainty model based on Ellis et al. (2000a) was used to quantify uncertainty in landscape change analysis. First, each map class area estimate \( A_k \) was described using a normal probability distribution function (PDF) with \( \mu = \) class area (the sum of polygon areas within each class) and \( \sigma = \) model-estimated error in map class area. To incorporate error of omission, an “omitted class” area estimator \( O \) was prepared, based on a 10% overall error of omission (Ellis & Wang, submitted for publication), specified as a binomial PDF with \( n = 1 \) and \( p = 0.10 \) multiplied by a normal PDF with \( \mu = 0 \) and \( \sigma = 10\%/Z_{0.05} \) \((Z_{0.05} = 1.645)\). Total site area was specified as a normal PDF with \( \mu = 100\% \) and \( \sigma = 0.05\%/Z_{0.05} \) \((0.05\% = \) the error tolerance for total mapped area). To prevent negative area values beyond the error tolerance for total mapped area, the lower bound of all area PDFs \( (A_k, O, A_t) \) was truncated to −0.05% of site area.

In separate MC models prepared for each site, individual map class area estimates \( A_k \) were divided by the sum of all map class area estimates \( (n \) classes) plus the “omitted class” area \( O \), and multiplied by total site area \( A_t \) to produce normalized map class area estimates \( CA_k \):

\[
CA_k = \frac{A_t}{n} \frac{A_k}{\sum_i A_k + O}
\]

(1)

These normalized area estimates include error of omission, are correlated with other map class areas and sum to the total area of each site (within the error of measurement), incorporating the basic error structure of area measurements from thematic mapping. Uncertainty in area changes was calculated by subtracting historical from current map class area estimates \( CA_k \) using MC uncertainty analysis (Ellis et al., 2000a). When a map class was present only in one time period, a “zero estimator” was specified for the missing class area using the same formula as for \( O \), above, but with \( \sigma \) set to the error for the time period in which the class was present.

Data quality of all estimates was described using ordinal “grades” from \( \{0\} \) to \( \{4\} \), with grades of \( \{3\} \) and above representing direct measurements and those below incorporating significant subjectivity (Ellis et al., 2000a). Data quality of each ecotope feature was graded directly by interpreters; grades for maps and map classes were calculated by feature-area-weighted averaging. Grades were reduced by one unit for all map classes smaller than 0.25% of site area, as these were not reliably classified (Ellis & Wang, submitted for publication).

Grades for area changes were calculated using a rule-based data quality algorithm (Ellis et al., 2000a).

Ecotope to ecotope transformations and landscape changes within sites were visualized by intersecting current and historical maps using GIS to create “change feature” maps. Data quality of each change feature was set equal to the minimum value across time periods, and then reduced by one unit for all features falling within the “potential false change zone” of maps, as described by buffering the perimeters of change features to the error in image coregistration within each site (CE90 at Independent Check Points, Table 5 of; Wang & Ellis, 2005a). Changes in human impact were highlighted by an ordinal USE change index (change in USE class rank in Table 3: 0 to 14, bottom to top; classes ranked subjectively based on anthropogenic alteration of soils, vegetation and hydrology) with positive values indicating increased impact (maximum = 14 for change from Pristine to Constructed). Changes in potential vegetation growth were highlighted by a COVER change index (change in COVER class rank in Table 4: 0 to 7, bottom to top), with positive values indicating decreased potential for vegetation growth (maximum = 7 for change from Perennial to Sealed).

3. Results

3.1. Changes in fine-scale ecological features

Ecotope maps revealed fine-scale landscape heterogeneity within all sites and demonstrated increases in this heterogeneity over time (Fig. 2, Table 5). The total number, perimeter and size of ecotope features varied greatly, but the median size of ecotope features was always small, <0.2 ha within all sites, with a cross-site median of 520 m², explaining the large number and extensive edges of features in all sites (Table 5). The diversity of ecologically distinct anthropogenic landscape features was evident in the 226 unique ecotope classes observed across sites. However, only 149 of these ever covered >0.25% of any site, the minimum required for reliable classification (Ellis & Wang, submitted for publication) and 36 was the maximum reliably classified within any 1 km² site, out of 57 observed (Table 5). Reliable ecotope classes covered >97.8% of site area within all sites (Table 5), so that map data quality was consistently high \((\{4\} \) for current, \( \{3\} \) for historic maps), except for a \( \{3\} \) in 2003 Cub Hill, where access for field validation was limited, and \( \{2\} \) in 1944 Jintang where image quality was poorest. Area measurement error varied more widely, with a maximum in Baltimore where the predominance of small features pushed the perimeter-based error model to its limit (Table 5; Ellis & Wang, submitted for publication).

The overwhelming importance of fine-scale ecological changes within anthropogenic landscapes was confirmed by the observation that land COVER changed by 21% to 52% across 1 km² sites, yet the median area of contiguous COVER changes was always less than 260 m², and most COVER change occurred in patches smaller than 0.4 ha in all sites, and in patches <0.1 ha in two sites (Table 6). Sites were increasingly fragmented over time, as demonstrated by increased ecotope feature count, perimeter, and COVER patch density and also by decreased median feature size at all sites except Gaoyi (Table 5). Current maps always had substantially more ecotope classes...
than historic maps (except in Cub Hill; Table 5), demonstrating increased landscape complexity as well.

3.2. General differences between sites

Patterns of land FORM, USE, and COVER varied widely across sites, highlighting broad differences in population and environment (Fig. 3, Table 1). Constructed and Disturbed USE were ubiquitous, but agriculture dominated all sites except Baltimore, from the simple irrigated village plain of Gaoyi to the more complex mixtures of Paddy, Forestry, and Rainfed agriculture at other sites (Fig. 3b). The transformation of Cub Hill from rural to suburban was characterized by a dramatic increase in Ornamental and Constructed USE; apart from a small area in Gaoyi, Ornamental USE was only present in US sites. This simple characterization of sites was also captured by the higher USE index values for Baltimore, current Cub Hill and Gaoyi (Table 5).
Population density explained much of the variability between sites, and also highlighted relationships between population, impervious surfaces, and landscape fragmentation. Site population was tightly linked with sealed COVER ($R^2 = 0.81$, $P < 0.001$), COVER index ($R^2 = 0.76$, $P < 0.001$) and COVER patch density ($R^2 = 0.65$, $P < 0.01$), and was a modest predictor of feature count and perimeter ($R^2 > 0.39$; $P < 0.05$), demonstrating linkages between population increase, impervious surfaces, and landscape fragmentation. By contrast, none of the environmental factors in Table 1 was significantly related to site COVER or to the ecotope map characteristics in Table 5.

3.3. Changes in land USE and COVER

Long-term changes in USE and COVER demonstrated a general trend toward increasing human impact and increased perennial vegetation across sites over time (Fig. 4). Constructed USE increased significantly across all sites except Baltimore, where it decreased in parallel with Sealed COVER. Disturbed USE also increased significantly across sites. Significant declines in Annual COVER across Cub Hill, Jintang, and Dianbai were balanced by increased Perennial and/or Mixed COVER, except in Gaoyi, where Annual COVER was replaced by Sealed COVER. Annual and Mixed COVER replaced Sealed COVER in Baltimore, while in Yiyang, uncertainty obscured the source of a significant decline in Mixed COVER.

These general patterns of change were highlighted within and across sites by USE and COVER change indices (Figs. 5 and 6, Table 5). Changes were finely dispersed across urban Baltimore, except for a corridor of change along a new highway, in contrast with the complete transformation of western Cub Hill to a residential suburb. Fine-scale changes were associated with housing, irrigation, roads and other anthropogenic structures in China sites, in clumped patterns where houses were built near each other (Gaoyi and Dianbai) and along the edges of hills in Jintang and Yiyang where houses were dispersed. These built-up areas were generally associated with the regeneration of perennial vegetation, including the transformation of agricultural land into woody and tree-covered yards in Cub Hill, the conversion of buildings to grassy highway medians in Baltimore, and the regrowth of vegetation in Disturbed areas around newly constructed houses in China. The widespread presence of simultaneous fine-scale increases and decreases in human impact and vegetation growth within and across sites further demonstrates the ecological complexity of long-term anthropogenic change. At the site level however, USE indices yielded simple results, indicating net increases in human impacts in Cub Hill and in

### Table 5

<table>
<thead>
<tr>
<th>Ecotope features</th>
<th>Baltimore</th>
<th>Cub Hill</th>
<th>Gaoyi</th>
<th>Jintang</th>
<th>Dianbai</th>
<th>Yiyang</th>
<th>Median</th>
</tr>
</thead>
<tbody>
<tr>
<td>Feature areas (m$^2$)</td>
<td>173</td>
<td>180</td>
<td>65</td>
<td>131</td>
<td>39</td>
<td>81</td>
<td>67</td>
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<tr>
<td>Median</td>
<td>475</td>
<td>334</td>
<td>377</td>
<td>265</td>
<td>219</td>
<td>497</td>
<td>1330</td>
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<td>5 percentile</td>
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<td>47</td>
<td>53</td>
<td>44</td>
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<td>95 percentile</td>
<td>2396</td>
<td>1968</td>
<td>21333</td>
<td>5371</td>
<td>46503</td>
<td>7739</td>
<td>14365</td>
</tr>
</tbody>
</table>

**Indices**

- USE index
- COVER index
- COVER patch density (km$^{-2}$)
- COVER patch median area (m$^2$)

**Ecotope classes**

- Totala
- Reliable classesb
- Reliable area (%)c
- Area error (%)d

<table>
<thead>
<tr>
<th>Ecotope features</th>
<th>Total</th>
<th>Reliable classes</th>
<th>Reliable area (%)</th>
<th>Area error (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>USE index</td>
<td>13</td>
<td>6</td>
<td>98.8</td>
<td>40</td>
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<tr>
<td>COVER index</td>
<td>6</td>
<td>2</td>
<td>98.4</td>
<td>23</td>
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<tr>
<td>COVER patch density</td>
<td>343</td>
<td>2</td>
<td>98.3</td>
<td>22</td>
</tr>
<tr>
<td>COVER patch median area (m$^2$)</td>
<td>272</td>
<td>2</td>
<td>98.0</td>
<td>11</td>
</tr>
</tbody>
</table>

### Table 6

<table>
<thead>
<tr>
<th>Land COVER change in patches &gt;25 m$^2$ after conversion to a 2 m grid, circa 1950 to 2002</th>
</tr>
</thead>
<tbody>
<tr>
<td>Baltimore</td>
</tr>
<tr>
<td>Site area (%)</td>
</tr>
<tr>
<td>Count</td>
</tr>
<tr>
<td>Median area (m$^2$)</td>
</tr>
<tr>
<td>Patches &lt;0.1 ha (%)</td>
</tr>
<tr>
<td>Patches &lt;0.4 ha (%)</td>
</tr>
<tr>
<td>Patch density (%)</td>
</tr>
</tbody>
</table>

---

*a* Percent of COVER change occurring in patches smaller than 0.1 and 0.4 ha.  
*b* Change in COVER patch density (current–historical, in km$^{-2}$) as a percentage of current patch density.
China sites, with no net change in Baltimore, while the COVER index pointed only toward robust changes in Sealed COVER, with a decrease in Baltimore, an increase in Cub Hill and Gaoyi, and no change elsewhere (Table 5).

3.4. Ecotope-level analysis

Ecotope-level analysis further highlighted the ecological complexity of densely populated landscapes and their changes (Table 7, Figs. 7 and 8). The extent of each site covered by the ten largest ecotopes, illustrated by bar length in Fig. 7, revealed a tendency toward greater numbers of smaller ecotopes over time, as was also apparent from total ecotope number (Table 5). The trend toward increasing landscape complexity was also evident in ecotope area changes, where major decreases in the largest ecotopes were generally not matched by equally large increases in other ecotopes (Fig. 8a). Indeed, in three sites, the five largest ecotope transformations were merely the division of the site’s largest ecotope into five smaller ecotopes (Fig. 8b). The ten largest ecotope transformations accounted for between 29% and 74% of total changed area across sites, indicating simpler land transformations where this indicator was largest (Gaoyi, Jintang) and more complex transformations where it was smallest (Baltimore, Cub Hill), with the total number of ecotope transformations following the same trend (Table 7). Given that the number of ecotope transformations needed to account for 90% of changed area was far greater than the number measured reliably, a large number of relatively small
and unreliably measured ecotope transformations are needed to explain as little as 90% of change within any site (Table 7). This uncertainty was exacerbated by the 43% to 83% of site changed area that was within the portion of maps susceptible to false change errors caused by the misregistration of maps (Table 7).

Detailed ecotope-level analysis was needed to explain the causes and consequences of even the largest changes observed across sites (Table 1, Figs. 3–8). Urban Baltimore’s exceptional decline in population and impervious surfaces was caused by the replacement of row houses with a highway containing a wide grass median and by increases in parks, abandoned lots, yards, and playgrounds. Suburban Cub Hill, where population increase and landscape transformation were greatest, changed primarily by the conversion of annual crops and pasture to woody and tree-covered yards around newly constructed houses and roads. Remarkably, the site’s largest ecotope transformations were not increases in housing and roads, but changes from annual crops to pastures, from pastures and deciduous forests to woody yards, and from coniferous to deciduous forest, which explains why the site’s suburban development actually increased perennial vegetation cover. Gaoyi’s population also more than doubled, but its land transformation was the smallest and simplest across sites, with nearly all change caused by a brick factory and village expansion into agricultural land, accompanied by tree growth around houses and roads. Jintang’s increase in agricultural population was associated with increased perennial vegetation cover, mostly by improved forestry on Steep Slopes and by introduction of citrus orchards to Bench Plateaus, and was also accompanied by a change from reservoir paddy (winter fallow) to regular rice paddy management, allowing crop production in winter. Dianbai’s population expanded into agricultural land around two small villages, helping to explain the site’s exceptional loss of Annual COVER, but the main cause of this was the introduction of litchi orchards and forestry into staple crop areas and hilly areas where shrubby vegetation was maintained historically by fuel gathering and periodic burning. Yiyang, the least populated site, also changed the least in terms of land use and cover, despite a large area experiencing ecotope change, mainly by changing from coniferous forest in hilly areas to a mix of closed canopy woody vegetation, mixed forest, and marginal tea production.

4. Discussion

4.1. The global importance of fine-scale ecological change

The results of this study confirm that fine-scale ecological changes within densely populated landscapes are both abundant and complex, and that the complexity of these landscapes has generally increased over time. Though land cover changed substantially across every site, by 20% to 50% of total site area, most of this change occurred at very fine scales, in patches smaller than 0.4 ha in all sites, and in patches smaller than 0.1 ha in two sites (Table 6, Fig. 6). These fine-scale changes are too small for precise measurement by conventional land-cover mapping methods, especially those depending on ≥ 30 m (~0.1 ha) pixels (Forster, 1985; Woodcock & Strahler, 1987). In terms of 1 km resolution global land cover, no change at all would be measured across China sites (“croplands”) or in Baltimore (“urban and built-up”), though Cub Hill’s transformation from agricultural to suburban would be observable as a change from “cropland/natural vegetation mosaic” to “urban and built-up” (e.g. Hansen et al., 2000). These comparisons highlight the difference between extensive landscape transformation processes, such as deforestation or urban expansion, that are well measured by conventional methods, and the intensive landscape transformation processes that occur within densely populated anthropogenic landscapes after their creation by extensive transformation processes. Given that urban areas cover nearly 3 million km² globally (Center for International
Earth Science Information Network (CIESIN) et al., 2004) and densely populated agricultural villages cover more than 6 million km² of Asia (Ellis, 2004), the global importance of intensive landscape transformation processes is clear.

Fine-scale change measurements across a range of anthropogenic landscapes confirmed some basic assumptions about intensive landscape transformation processes while refuting others. As expected, anthropogenic impervious surface areas increased with population and therefore increased at all sites except urban Baltimore, where population decreased over time. In three of six sites, including one in rural China, current impervious surface area was well above the 10% threshold implicated in serious environmental degradation (Paul & Meyer, 2001). Human population density was also strongly linked with landscape fragmentation, potentially threatening biodiversity (Saunders et al., 1991). In contrast, at all five historically agricultural sites, rainfed staple crop agriculture declined substantially as population increased, driving a widespread decline in herbaceous vegetation cover. The pathways of this change differed among sites, ranging from housing development, conversion to perennial crops, and the expansion of more intensive agricultural systems including intensive cropping, greenhouses, irrigated agriculture and paddy rice production.

The most unexpected result of this study was that population increases in the already densely populated agricultural landscapes of China and the USA were not associated with a net decline in perennial cover, but rather with substantial increases in this cover at four of five historically agricultural sites. As perennial vegetation, itself a carbon sink, is also associated with higher soil carbon content, this result indicates that Asia’s densely populated agricultural landscapes may represent a globally significant carbon sink (Heilig, 1994; Houghton, 2003; Rudel et al., 2005). The causes of this perennial recovery

Fig. 5. Changes in USE index across 1 km² sites, circa 1950 to 2002. Baltimore (a), Cub Hill (b), Gaoyi (c), Jintang (d), Dianbai (e), Yiyang (f). Legend illustrates USE index changes indicating full intensification (index values 5 to 14), increase (1 to 4), reduction (−4 to −1), and recovery from human impact (−14 to −5).
differed significantly among sites, ranging from the abandonment of annual cropland followed by perennial regrowth, the planting and regrowth of trees around houses, improved and expanded forestry management, and the introduction of orchard crops both within existing agricultural areas and across relatively denuded hillsides. This wide variety of landscape transformation

Table 7
Ecotope changes across sites, ~1950 to ~2002

<table>
<thead>
<tr>
<th></th>
<th>Baltimore</th>
<th>Cub Hill</th>
<th>Gaoyi</th>
<th>Jintang</th>
<th>Dianbai</th>
<th>Yiyang</th>
<th>Median</th>
</tr>
</thead>
<tbody>
<tr>
<td>Changed area (% of site)</td>
<td>43</td>
<td>87</td>
<td>28</td>
<td>63</td>
<td>67</td>
<td>65</td>
<td>64</td>
</tr>
<tr>
<td>Top 10 transitions</td>
<td>29</td>
<td>38</td>
<td>74</td>
<td>56</td>
<td>40</td>
<td>50</td>
<td>45</td>
</tr>
<tr>
<td>Reliable</td>
<td>9</td>
<td>50</td>
<td>5</td>
<td>18</td>
<td>28</td>
<td>33</td>
<td>23</td>
</tr>
<tr>
<td>Ecotope transformations</td>
<td>460</td>
<td>456</td>
<td>112</td>
<td>302</td>
<td>230</td>
<td>276</td>
<td>289</td>
</tr>
<tr>
<td>90% of change</td>
<td>169</td>
<td>114</td>
<td>27</td>
<td>83</td>
<td>70</td>
<td>77</td>
<td>80</td>
</tr>
<tr>
<td>Significant</td>
<td>36</td>
<td>69</td>
<td>16</td>
<td>48</td>
<td>58</td>
<td>43</td>
<td>45.5</td>
</tr>
</tbody>
</table>

*Percent of site with a change in ecotope class, changed by the top 10 ecotope to ecotope transitions, and reliable changed area (changes outside the potential false change zone).

*Total unique ecotope to ecotope transitions, the smallest number needed to account for ≥90% of changed area, and the number of transitions with area >0.25% of site area.
Fig. 7. The ten largest ecotope classes within each site. Ecotope classes are sorted by area, symbolized by USE class (legend) and labeled using standard ecotope codes based on land FORM (Table 2), USE (Table 3), COVER (Table 4), and GROUP+TYPE classes. Bars are proportional to site cover, with 100% indicated by a bracket beneath each site label.
pathways helps explain why site population and environmental factors were not significant predictors of land-cover changes across sites, except for impervious cover, and indicates that these changes are controlled by factors not considered, such as economics and policy — a useful avenue for further study.

4.2. Costs and benefits of measuring fine-scale ecological change

Recent developments in remote sensing, including high spatial resolution multi- and hyper-spectral imaging, spectral unmixing, Light Detection and Ranging (LIDAR), automated feature extraction, enhanced stereo editing, and other techniques will certainly enhance fine-scale ecological change measurements in the future (e.g. Hill et al., 2002; Hurtt et al., 2004; Goetz et al., 2003; Sawaya et al., 2003; Small, 2003; Tao et al., 2004). However, the mapping of land management practices in detail will always depend on local fieldwork, and data obtained by the newest technologies are rarely compatible with the historical data needed to measure long-term change. On the other hand, anthropogenic ecotope mapping is expensive, requires intensive training, fieldwork, local assistance, and high resolution imagery, and is limited by the extent of historical imagery and elders capable of map validation. The methodology is also best suited to densely populated areas with intensive land management — the opposite of conventional methods for land-cover mapping (Forster, 1985; Woodcock & Strahler, 1987; Smith et al., 2003).

Fig. 8. Ecotope changes within sites, circa 1950 to 2002. (a) The top five decreases (negative; top) and top five increases (positive; below) in ecotope class areas as a percent of site area are presented as bars symbolized by USE class, with 90% confidence intervals. (b) The top five ecotope to ecotope transformations within each site (past—present ecotope transitions), presented as a percent of total site area. Ecotope labels are described in Fig. 7.
High spatial resolution remotely sensed measurements are increasingly being integrated into regional and global environmental monitoring programs, including those for urban areas (e.g. Achard et al., 2002; Cihlar et al., 2000; Morisette et al., 2003; Small, 2005). In densely populated rural, urban and suburban landscapes, anthropogenic ecotope mapping complements these measurements by detailing the complex anthropogenic causes and ecological consequences of fine-scale change processes that dominate these landscapes. For example, ecotope-level analysis revealed the transformation of woody pasture to tree-covered yards by suburban development in Cub Hill, the conversion of abandoned houses to woody vacant lots following population decline in urban Baltimore, the end of winter paddy flooding in Jintang by improved irrigation in the 1970s, and increases in perennial vegetation cover in Jintang and Dianbai brought about by a combination of reforestation programs in the 1980s and the expansion of orchards in the 1990s. All of these changes were driven by national policy and regional economics interacting with local land managers. All of these have undoubtedly produced regional consequences in terms of biogeochemistry and biodiversity (Saunders et al., 1991; Li et al., 2002). Yet the causes and consequences of these changes are not revealed simply by measuring changes in perennial, flooded, or impervious cover.

Anthropogenic ecotope mapping supports generalized change comparisons within and among sites and regions and includes a change indexing system that helps distinguish ecologically significant long-term changes from minor fluctuations in land use and cover (Table 5, Figs. 5 and 6). Once more general changes have been observed, the more detailed and comprehensive ecological information available in the ecotope classification system is critical for investigating the drivers of these changes and their impacts. For example, the combination of different USE classes, such as Forestry or Fallow, with the same GROUP class, “deciduous broadleaf trees”, adds information on land management practices with profoundly different ecological impacts, while different FORM classes, such as Steep Slope vs. Floodplain, incorporate critical differences in edaphic environment that control biogeochemical processes such as erosion and denitrification. Most importantly, anthropogenic ecotope mapping provides a platform for more detailed investigations of ecological changes in densely populated landscapes, yielding practical sample strata for field measurements on soils, vegetation, and other parameters, and serving equally well as land survey units when obtaining land management data directly from managers in the field (Ellis, 2004). Finally, the methodology incorporates a robust uncertainty analysis system that enables integration of these different data types to make statistically reliable long-term ecological change estimates (Ellis et al., 2000a,b).

4.3. Conclusions

This study presents the first standardized high spatial resolution measurements of long-term ecological changes across the full range of densely populated anthropogenic landscapes, from rural, to suburban, to urban. The complexity and ecological significance of the fine-scale changes revealed by these observations, including simultaneous increases in impervious surface area and perennial vegetation cover, confirm the need to measure these across densely populated landscapes if we are to understand and mediate their local, regional and global consequences. This can be accomplished by integrating high spatial resolution ecological change measurements, such as those produced by anthropogenic ecotope mapping, into regional and global environmental monitoring programs by means of regionally stratified sampling designs across densely populated regions. By this approach, global and regional data from remote sensing can be linked with samples of fine-scale ecological change measurements, integrating landscape changes with direct measurements on ecological parameters, such as vegetation biomass and soil carbon, and the local practices of land managers, including biomass combustion and fertilizer application. This will allow more accurate and spatially explicit inventories of carbon sequestration and other globally important ecological processes than is currently achievable by plot or regional methods alone. As densely populated landscapes increase their global extent in the years ahead, primarily by urban expansion, the global importance of long-term ecological changes in anthropogenic landscapes will only increase, along with the need to measure, understand, and mediate their more negative consequences.

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References


