Spatial and temporal changes in Aniene river basin (Latium, Italy) using landscape metrics and moving window technique

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Abstract
Landscape ecology is largely founded on the notion that environmental patterns strongly influence ecological processes and anthropogenic activities can disrupt the structural integrity of landscapes and change ecological flows across the landscape. For these and other reasons have been developed methods to quantify landscape patterns. This study focused on Aniene river basin changes both at spatial and temporal scale. The results highlights a good stability of the woodland patches closer to the mountain chains ridges and in the middle of the river basin and losses of the forest patches closer to the valleys areas and in the periphery of Rome.

Keywords: river basin, change detection, moving Windows, landscape indices.

Introduction
Landscape ecology, if not ecology in general, is largely founded on the notion that environmental patterns strongly influence ecological processes [Turner, 1989; Gardner and...
O’Neill, 1991; Turner et al., 2001]. The habitats in which organisms live, for example, are spatially structured at a number of scales, and these patterns interact with organism perception and behavior to drive the higher level processes of population dynamics and community structure [Johnson et al., 1992]. Anthropogenic activities can disrupt the structural integrity of landscapes and is expected to impede, or in some cases facilitate, ecological flows (movement of organisms) across the landscape [Gardner et al., 1993]. A disruption in landscape patterns may therefore compromise its functional integrity by interfering with critical ecological processes necessary for population persistence and the maintenance of biodiversity and ecosystem health [With and King, 1999]. For these and other reasons, much emphasis has been placed on developing methods to quantify landscape patterns, which is considered a prerequisite to the study of pattern-process relationships [O’Neill et al., 1988; Turner, 1990; Turner and Dale, 1991; Baker and Cai, 1992; McGarigal and Marks, 1995]. This has resulted in the development of literally hundreds of indices of landscape patterns. Definitions of landscape invariably include an area of land containing a mosaic of patches or landscape elements. Forman and Godron [1986] defined landscape as a heterogeneous land area composed of a cluster of interacting ecosystems that is repeated in similar form throughout. The concept differs from the traditional ecosystem concept in focusing on groups of ecosystems and the interactions among them. Landscapes do not exist in isolation. Landscapes are nested within larger landscapes, that are nested within larger landscapes, and so on. In other words, each landscape has a context or regional setting, regardless of scale and how the landscape is defined. The landscape context may constrain processes operating within the landscape. Landscapes are “open” systems; energy, materials, and organisms move into and out of the landscape. This is especially true in practice, where landscapes are often somewhat arbitrarily delineated. That broad-scale processes act to constrain or influence finer-scale phenomena is one of the key principles of hierarchy theory [Allen and Star, 1982; Odum et al., 2006] and ‘supply-side’ ecology [Roughgarden et al., 1987]. The importance of the landscape context is dependent on the phenomenon of interest, but typically varies as a function of the “openness” of the landscape. The “openness” of the landscape depends not only on the phenomenon under consideration, but on the basis used for delineating the landscape boundary. For example, from a geomorphological or hydrological perspective, the watershed forms a natural landscape, and a landscape defined in this manner might be considered relatively “closed”. Of course, energy and materials flow out of this landscape and the landscape context influences the input of energy and materials by affecting climate and so forth, but the system is nevertheless relatively closed. Conversely, from the perspective of a bird population, topographic boundaries may have little ecological relevance, and the landscape defined on the basis of watershed boundaries might be considered relatively “open” system. Local bird abundance patterns may be produced not only by local processes or events operating within the designated landscape, but also by the dynamics of regional populations or events elsewhere in the species’ range [Wiens, 1981, 1989; Vaisanen et al., 1986; Haila et al., 1987; Ricklefs, 1987]. Landscape metrics quantify the pattern of the landscape within the designated landscape boundary only. Consequently, the interpretation of these metrics and their ecological significance requires an acute awareness of the landscape context and the openness of the landscape relative to the phenomenon under consideration. The common usage of the term “landscape metrics” refers exclusively to indices developed for categorical
map patterns. Landscape metrics are algorithms that quantify specific spatial characteristics of patches, classes of patches, or entire landscape mosaics. A plethora of metrics has been developed to quantify categorical map patterns. These metrics fall into two general categories: those that quantify the composition of the map without reference to spatial attributes, and those that quantify the spatial configuration of the map, requiring spatial information for their calculation [McGarigal and Marks, 1995; Gustafson, 1998].

Composition is easily quantified and refers to features associated with the variety and abundance of patch types within the landscape, but without considering the spatial character, placement, or location of patches within the mosaic. Because composition requires integration over all patch types, composition metrics are only applicable at the landscape-level. Spatial configuration is much more difficult to quantify and refers to the spatial character and arrangement, position, or orientation of patches within the class or landscape. Often the values of the landscape ecology indices changes suddenly with the hierarchical scale of resolution so that to study the differences among different scale are necessary implemented techniques of analysis. Remote sensing of earth surface really changes radically the ecologist’s perspective of landscapes, watershed and in general of any natural object on earth surface. The contribution of Remote Sensing to modern applied ecology, landscape ecology and land management has been relevant in the last ten years [Ales et al., 1992; Allen, 1994; Frohn, 1998; Mertes, 2002]. The Landsat TM imagery has got an useful resolution for ecological research at landscape level and has been used to detect a great variety of ecological systems like urban and agricultural aggregates [Fuller et al., 1994], river basins [Michener and Houholuis, 1997; McGuire et al., 1990], herbaceous vegetation [Joria and Jorgensen, 1996], forests and deforestation [Tucker et al., 1984], river basin [Mertes, 2002]. The changes that can occurred in a watershed are linked to the land use practices: the landscape act like a “sink” because of its function of retention of nutrients that modulates water quality [Johnston et al., 1997] so, landscape and river has to be considered an unique interacting ecosystem [Haslam, 2006]. Remote sensing offers important tools like classification procedures and change detection analysis for the monitoring of both land use and temporal changes occurring in the landscape. Many change detection investigations tend to focus on general description of the vegetation and may present satellite data as a qualitative tool for studying specific ecosystems [Narumalani et al., 2004]. Rainis [2003] suggests that in addition to studying the composition of land use types, their spatial distribution and arrangement also need to be considered for monitoring changes. Landscape metrics can be used for this aim and can be derived for one of three levels: patch level (defined for individual patches), class level (characteristics of all patches in a given class), and the landscape level (integrated over all patch types or classes over the extent of the data). In this study has been used remote sensing to quantify land use/land cover changes, and landscape metrics to track ecological impacts on Aniene river basin. To detect landscape metric changes at local scale we used the moving window analysis who is implemented in fragstas 3.3 software. The moving window approach consists in a virtual squared or radial window who’s dimensions (radius or side) area determined a priori by the researcher on the basis of the smallest patch size in the map of and the objectives of the study. The virtual window is passed on the map by the software that simultaneously compute metrics in each window. The results are a series of images showing metrics values in the selected areas. In this The landscape metrics used has been chose on the basis of maximum information and
minimum redundancy criterion (avoid the patch level metrics) [Forman, 1995] and selecting those metrics useful to assess river network integrity.

Study area

The study site is the Aniene river basin, a large watershed, or riverscape (1414 Km²) extended on the left side of Tiber river. The Aniene river born in the Simbruini mountain chain and flows through the north side of Prenestini mountain chain and the volcanic complex of Colli Albani. Downstream the river reach the periphery of Rome and the village of Tivoli, than inflow in Tiber river (Fig.1). The riverscape is heterogeneous without a recognized matrix: crop fields, olives trees, orchard and vineyards are homogeneously distributed. Woodlands and natural reserves are well preserved, urban centres and industrial installation although presents are not dominant and limited to the downstream. Close to the metropolitan area of Rome the complexity and the fragmentation of the riverscape growths in function of the intensive agricultural practice mixed to urban periphery and factories and the river became the only corridor among the natural reserve in the middle course and the anthropic aggregates of the valley.

Figure 1- Study area.
Materials and Methods
To detect land use changes in the study area has been used two remote sensed imagines with a temporal distance of 21 years. The Landsat 5 TM image was acquired in August the 14th 1985, the Aster image was acquired in July 14th 2006. The Aster’s image visible bands have been resampled to modify their spatial resolution from 15 meters to 30 meters in order to obtain the required spatial coherence with the Landsat image. The imagines have been pre-processed: orthorectified, corrected for atmosphere scattering (dark object subtraction) and co-registered (UTM WGS 84 zone 33 North). To categorized the images we selected a pool of training regions (ROI, Region Of Interest) representative of riverscape land use, afterwards each ROI has been statistically validate for spectrum separability by elaboration of the apposite indices of Jeffries-Matusita and Transformed Divergence: both this measures have a range values from 0 to 2.0 and indicate how well the selected ROI pairs are statistically spectrally separated. Values greater than 1.9 indicate a high separability while ROI with very low separability values (less than 1.0) must be combined into a single ROI. Starting from the initially selected ROI and using the indices range values have been eliminated or combined those ROI who were not clearly spectrally separated. In the following step the images have been categorized using Maximum Likelihood (supervised classification procedure). Classifications have been validate elaborating a Confusion Matrix from ground true ROI relieved on field (Aster 2006: Overall accuracy 0.87 (87%), Kappa Coefficient 0.77, Landsat 1985, Overall accuracy 0.79 (79%), Kappa coefficient 0.69). The change detection analysis has been computed after classifications using the pixel by pixel difference [Singh, 1989] and normalized by using the algorithm proposed by Zurlini et al. [2006].

\[ D(x, y) = \left| \frac{f_{\tau_1}(x, y) - f_{\tau_2}(x, y)}{\sqrt{s_{\tau_1}^2 + s_{\tau_2}^2 - \text{cov}_{\tau_1\tau_2}}} \right| \]

where \( D(x, y) \) represents the standard of change, \( f_{\tau_1}, f_{\tau_2} \) are the measures for the pixel of coordinates \((x, y)\) and reported it to the image acquired at time \( \tau_1 \) time \( \tau_2 \), \( m \) is the mean value of the difference image, \( s_{\tau_1}^2 \) and \( s_{\tau_2}^2 \) represent the variance relative to time \( \tau_1 \) \( \tau_2 \) and \( \text{cov}_{\tau_1\tau_2} \) the covariance. Using this procedure the pixels follows a symmetrical normal distribution: pixels with a null variation are distributed around a mode equal to 0 and pixel with positive (increase) or negative (decreases) respectively, distributed in the right and left tail of the distribution. The statistical output of change detection is a matrix computed with a cross-tabulation algorithm. In the matrix the initial state classes are in the columns and the final state classes in the rows. The class changes row indicates the total number of initial state pixels that changed classes, the image difference row is the difference in the total number of equivalently classed pixels in the two images, computed by subtracting the initial state class totals from the final state class totals. An image difference that is positive indicates that the class size increased. Even though no pixels classified changed classes, the values of image difference can grew significantly. Values can be computed in pixels or in percent. The classified images have been exported in ASCII and GRID formats to be analyzed using an ad hoc set of landscape metrics. The following landscape metrics have been computed:
1) "Cohesion index (COHES)." Cohesion measures the physical connectedness of the corresponding patch type, it equals 1 minus the sum of patch perimeter (in terms of number of cells) divided by the sum of patch perimeter times the square root of patch area (in terms of number of cells) for all patches in the landscape, divided by 1 minus 1 over the square root of the total number of cells in the landscape (multiplied by 100 to convert to a percentage) [McGarigal and Marks, 1998];

\[
Cohes = \left[ 1 - \frac{\sum_{i=1}^{m} \sum_{j=1}^{n} P_{ij}}{\sum_{i=1}^{m} \sum_{j=1}^{n} (P_{ij} / \sqrt{A})} \right] \left[ 1 - \frac{1}{\sqrt{A}} \right]^{-1} \tag{100} \tag{2}
\]

\[P_{ij} = \text{perimeter of patch } ij \text{ in terms of number of cell surfaces}, \quad A_{ij} = \text{area of patch } ij \text{ in terms of number of cells}, \quad A = \text{total number of cells in the landscape.}\]

2) "Connectance (CONNECT)." Connectance generally refers to the functional connections among patches. What constitutes a "functional connection" between patches clearly depends on the application or process of interest; patches that are connected for bird dispersal might not be connected for salamanders, seed dispersal, fire spread, or hydrologic flow. Connections might be based on strict adjacency (touching), some threshold distance, some decreasing function of distance that reflects the probability of connection at a given distance, or a resistance-weighted distance function. Then various indices of overall connectedness can be derived based on the pair-wise connections between patches, one such index, connectance, can be defined on the number of functional joining. This index can be detected at only at landscape level. In the elaboration phase is possible to select the physical distance in meters at which detect connectance, defined on the number of functional joining between patches of the corresponding patch type [McGarigal and Marks, 1998];

\[
Connect = \left[ \sum_{j=1}^{m} \sum_{k=1}^{n} \frac{C_{ijk}}{\sum_{i=1}^{n} (n_i (n_i - 1) / 2)} \right] \tag{3}
\]

\[C_{ijk} = \text{joining between patch } j \text{ and } k \text{ (0 = unjoined, 1 = joined) of the same patch type, based on a user-specified threshold distance}, \quad n_i = \text{number of patches in the landscape of each patch type } i.\]

3) "Interspersion and juxtaposition index (IJI)." The interspersion/juxtaposition index increase in value as patches tend to be more evenly interspersed in a "salt and pepper" mixture. These and other metrics are generated from the matrix of pair wise adjacencies between all patch types, where the elements of the matrix are the proportions of edges in each pair wise type. Interspersion, on the other hand, refers to the intermixing of patches of different types and is based entirely on patch (as opposed to cell) adjacencies. There are several different approaches for measuring contagion and interspersion. One popular index that subsumes both dispersion and interspersion is the contagion index based on the
probability of finding a cell of type i next to a cell of type j [Li and Reynolds, 1993];

\[ IJI = - \sum_{k=1}^{m} \left( \frac{e_{ik}}{\sum_{l=1}^{m} e_{il}} \right) \ln \left( \frac{e_{ik}}{\sum_{l=1}^{m} e_{il}} \right) \]  \[4\]

\( e_{ik} \) = total length (m) of edge in landscape between patch types (classes) i and k.
\( m \) = number of patch types (classes) present in the landscape, including the landscape border.

4) “Shannon diversity index (SHDI) is a landscape level index and has been calculated with moving window technique. The moving windows technique is helpful to monitoring the effects of local scale on indices. The SHDI is a landscape level index, it cannot be calculated at class level”;

\[ SHDI = - \sum_{i=1}^{m} p_i \ln (p_i) \]  \[5\]

\( P_i \) = proportion of the landscape occupied by patch type (class) i.

5) “Aggregation index (AI). Aggregation index is calculated from an adjacency matrix, which shows the frequency with which different pairs of patch types (including like adjacencies between the same patch type) appear side-by-side on the map. Aggregation index takes into account only the like adjacencies involving the focal class, not adjacencies with other patch types. In addition, in contrast to all of the other metrics based on adjacencies, the aggregation index is based on like adjacencies tallied using the single-count method, in which each cell side is counted only once”;

\[ AI = \frac{g_{ii}}{\max \rightarrow g_{ii}} \]  \[6\]

\( g_{ii} \) = number of like adjacencies (joins) between pixels of patch type (class) i based on the single-count method and \( \max \rightarrow g_{ii} \) = maximum number of like adjacencies (joins) between pixels of patch type (class) i based on the single-count method.

6) “Patch density (PD) equals the number of patches of the corresponding patch type divided by total landscape area (m\(^2\))”;

\[ PD = \frac{n_i}{A} \]  \[7\]

\( n_i \) = number of patches in the landscape of patch type (class) I and \( A \) = total landscape area (m\(^2\)).
7) **Patch number** (PN) equals the total number of patches of the corresponding patch type (class).

### Moving window technique

The quantification of the changes in the patterns of disturbance at different scales is achieved by applying the methodology of the “moving window”, developed by Riitters et al. [2002] for the multiscale analysis of forest fragmentation. The **moving window** has been used in ecology for analysis fractal landscapes [Milne, 1991]. This approach consists in the construction of a virtual squared or radial window (fragstats 3.3. tool) whose dimensions (radius or side) are determined *a priori* by the operator on the basis of the smallest patch size of the map and the objectives of the study. The window of the specified shape and size is passed over every positively valued cell in the grid (all cells inside the landscape of interest). However, only cells in which the entire window is contained within the landscape are evaluated. Within each window, each selected metric at the class or landscape level is computed and the value returned to the focal (center) cell. Patch metrics are not allowed in the moving window analysis. The moving window is passed over the grid until every positively valued cell (including positively valued background cells) containing a full window is assessed in this manner. Because of the direct influence in river water quality of those patches closer to its course [Turner et al., 2001] and to prevent both the landscape edges effects and excesses of distances from river branches [McGarigal and Marks, 1995] in this study has been selected a squared window of 5 Km. The metrics has been calculated at class level except for the SHDI and CONNECT indices that can be calculated only at landscape level.

### Results and discussion

The change detection analysis and landscapes indices highlights some fundamental aspects of log-term structural dynamics of the study area in particular has been found a good stability of those woodland patches closer to the major mountain chains ridges and in the middle of the river basin, while there was a great variability in terms of losses of forested areas of those patches closer to the valleys and the periphery of Rome and Tivoli.

Woodlands patches appeared to be numerically increased in the considered lapse of time, passing from 10889 to 49089 patches, furthermore were found more disaggregates, with decrements in AI, COHES and JIJ respectively of -14.0%, -12.95% and - 0.30% (Tab. 1; Tab. 2). The total woodlands area resulted diminished with a quantifiable loss of – 45.881 pixels (Tab. 3). The PD was found increased as it is dependent on PN.

These results confirm the increase in fragmentation and the loss of connectivity among patches. Areas affected by the fragmentation and decrements of forest patches are those near the outskirts of Rome and Tivoli and the valleys areas while the mountain ridges and the neighbourhood areas were found not affected by the phenomenon (Fig 2).

In lowland areas, however there was a rise in monoculture. Crops patches resulted numerically decreased (32452 patches less in 2006 compared to 1985), furthermore Al, COHES and IJI indices were found increased, respectively+ 3.79%, +18.33% and+ 5.48%. The total crops area was found increased too (Tab. 3).
Table 1 - Main landscape indices elaborated for the classified Landsat TM5 1985 image. Class level calculation. Fragstats 3.3. [Mc Garigal and Marks, 1998].

<table>
<thead>
<tr>
<th></th>
<th>PN</th>
<th>PD</th>
<th>IJI 0-100</th>
<th>COHES 0-100</th>
<th>AI 0-100</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Industrial areas</strong></td>
<td>2288</td>
<td>14.074</td>
<td>75.45</td>
<td>53.78</td>
<td>32.18</td>
</tr>
<tr>
<td><strong>Woodlands</strong></td>
<td>10889</td>
<td>66.981</td>
<td>54.52</td>
<td>99.79</td>
<td>66.56</td>
</tr>
<tr>
<td><strong>Crops</strong></td>
<td>77752</td>
<td>478.273</td>
<td>60.01</td>
<td>80.58</td>
<td>47.02</td>
</tr>
<tr>
<td><strong>Orchards, Vineyards, Olive groves</strong></td>
<td>53568</td>
<td>329.511</td>
<td>57.62</td>
<td>48.28</td>
<td>27.35</td>
</tr>
<tr>
<td><strong>Grazing fields</strong></td>
<td>53309</td>
<td>327.918</td>
<td>55.63</td>
<td>62.20</td>
<td>34.71</td>
</tr>
<tr>
<td><strong>Urban areas</strong></td>
<td>23583</td>
<td>145.065</td>
<td>72.29</td>
<td>39.99</td>
<td>24.17</td>
</tr>
</tbody>
</table>

Table 2 - Main landscape indices elaborated for the classified Aster 2006 image. Class level calculation. Fragstats 3.3. [Mc Garigal and Marks, 1998].

<table>
<thead>
<tr>
<th></th>
<th>PN</th>
<th>PD</th>
<th>IJI 0-100</th>
<th>COHES 0-100</th>
<th>AI 0-100</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Industrial areas</strong></td>
<td>11722</td>
<td>72.105</td>
<td>71.16</td>
<td>63.28</td>
<td>33.63</td>
</tr>
<tr>
<td><strong>Woodlands</strong></td>
<td>49089</td>
<td>301.960</td>
<td>54.39</td>
<td>86.84</td>
<td>52.07</td>
</tr>
<tr>
<td><strong>Crops</strong></td>
<td>45300</td>
<td>278.652</td>
<td>65.49</td>
<td>98.91</td>
<td>50.81</td>
</tr>
<tr>
<td><strong>Orchards, Vineyards, Olive groves</strong></td>
<td>90101</td>
<td>554.235</td>
<td>57.96</td>
<td>71.97</td>
<td>38.74</td>
</tr>
<tr>
<td><strong>Grazing fields</strong></td>
<td>24540</td>
<td>150.952</td>
<td>65.98</td>
<td>54.29</td>
<td>30.90</td>
</tr>
<tr>
<td><strong>Urban areas</strong></td>
<td>26697</td>
<td>164.220</td>
<td>63.93</td>
<td>51.32</td>
<td>30.59</td>
</tr>
</tbody>
</table>

The main significance of these results is the shift from a direct agricultural practice to an intensive one. Orchards, vineyard and olive groves patches number was found increased (36533 new patches) and both COHES and AI were found incremented respectively +23.69% and +11.39%. The PD values of this class of land use increase as the total area: orchards, vineyards, olive grows area was found more than duplicate with change detection analysis. Change detection and landscape metrics indicates that the main changes occurred in the studied area consists in an extension and aggregation of patches: herbaceous crop fields become more extend than in the past replacing forests areas and crops trees replacing herbaceous crops as in a domino effect. Both these classes of land use furthermore resulted remodelled according to the intensive farming practices. In the industrial and urban areas both PN and PD indices values were found moderately increased. Industrial patches recorded an increase (94343 new patches) while urban areas patches expanded moderately (3114 new patches). The PD of these classes was found increased: +11.33% and +9.5% respectively. Has been recorded also a moderate rise in AI and COHES indices. Very important is the change detection statistical output: change detection matrix highlights a strong increment of industrial areas that were found more than quadrupled. Urban areas rise as well (+49.55%). On the basis of this results we can suppose there was a condensation of both new industrial installations and urban agglomerate around a pre-existing core: industrial installations areas however have occupied urban soils (and also partially crops areas), urban areas have mostly replaced crops in the outskirts of the cities. Particular attention should be address to the considerable reduction of grazing fields in terms of total areas (-78.67%) and class changes. A consistent part of grazing fields areas has been replaced by forests.
because of the ecological secondary succession phenomenon due to the abandonment of rough grazing practice.

Table 3 - Change detection matrix. Class changes and Image difference in percent. Initial state classes in the columns and the final state classes in the rows.

<table>
<thead>
<tr>
<th></th>
<th>Industrial areas</th>
<th>Water</th>
<th>Woodlands</th>
<th>Crops</th>
<th>Orchards, Vineyards, Olive groves</th>
<th>Grazing fields</th>
<th>Urban areas</th>
</tr>
</thead>
<tbody>
<tr>
<td>Industrial areas</td>
<td>46.51</td>
<td>0.85</td>
<td>2.46</td>
<td>1.04</td>
<td>0.86</td>
<td>0.72</td>
<td>4.23</td>
</tr>
<tr>
<td>Water</td>
<td>0.14</td>
<td>3.55</td>
<td>0.09</td>
<td>0.18</td>
<td>0.10</td>
<td>0.08</td>
<td>0.05</td>
</tr>
<tr>
<td>Woodlands</td>
<td>0.62</td>
<td>13.38</td>
<td>42.14</td>
<td>12.63</td>
<td>4.63</td>
<td>12.03</td>
<td>1.71</td>
</tr>
<tr>
<td>Crops</td>
<td>19.48</td>
<td>42.00</td>
<td>32.94</td>
<td>48.12</td>
<td>42.93</td>
<td>37.65</td>
<td>32.82</td>
</tr>
<tr>
<td>Orchards, Vineyards, Olive groves</td>
<td>7.87</td>
<td>30.71</td>
<td>15.23</td>
<td>30.76</td>
<td>43.30</td>
<td>27.45</td>
<td>25.80</td>
</tr>
<tr>
<td>Grazing fields</td>
<td>0.84</td>
<td>0.24</td>
<td>2.56</td>
<td>1.89</td>
<td>0.60</td>
<td>17.42</td>
<td>0.83</td>
</tr>
<tr>
<td>Urban areas</td>
<td>21.82</td>
<td>6.25</td>
<td>3.11</td>
<td>2.82</td>
<td>4.37</td>
<td>0.74</td>
<td>29.81</td>
</tr>
<tr>
<td>Class Changes</td>
<td>50.77</td>
<td>93.43</td>
<td>56.39</td>
<td>49.32</td>
<td>53.49</td>
<td>82.58</td>
<td>65.44</td>
</tr>
<tr>
<td>Image Difference</td>
<td>466.08</td>
<td>-38.55</td>
<td>-45.881</td>
<td>22.56</td>
<td>250.13</td>
<td>-78.67</td>
<td>49.55</td>
</tr>
</tbody>
</table>

This is particularly evident in the image difference: increment of woodlands are localized in slopes of the mountain and are mainly due to grazing fields–forests ecological successions (Fig. 2, white areas). Landscape metrics confirms the increment of fragmentation and the decrement of PN and PD (Tab.1; Tab. 2). The decrement of total water areas needs to be more investigates and supported with historical cartography and long term climate data. Any hypothesis can be done: over-draws from wells or long periods of drought that dried creeks but it must be supported by data.

Moving windows results
Moving windows analysis highlights differences among landscape and local scale indices values. High values of SHDI index (Fig. 3) were found in the middle course of the river
area (Fig. 3), This is a predominantly mountainous area with a good land use breakdown on respect to the lowlands (low values of the SHDI index).

Figure 2 - Change detection image difference of Aniene river watershed (Latium, Italy). Gray-scale colour: Stable areas = black; increments per class of land use = white coloured, decrements per class of land use = gray. Black areas corresponds to mountain chains ridges. Elaborated with pixel by pixel difference. ENVI 4.5.

Figure 3 – Shannon diversity index value at local scale. Moving window width 5km. Landscape level calculation. Hydrographic network overlayed.
Cohesion index values (Fig. 4) were found high in the areas closer to Rome and Tivoli: these two lowlands areas are tightly connected and corresponds to intensive crop fields cultivations. These evidences confirms the change detection analysis but highlights a better condition of those area localized in the middle of the watershed. Taking into account both SHDI and COHES values we can establish that the middle river course has got high diversity and good patches cohesion. The upstream areas shows low SHDI values and denotes a low heterogeneity in land-use. Upstream areas are prevalently hilly and localized close to the river spring but, conversely to be pristine are probably becoming altered due both to the increment of crop trees cultivation practices and the expansion of a lots of small anthropic agglomerates.

The CONNECT index (Fig. 5) values distribution highlights a good preservation of riparian vegetation: high values of the index are localized mainly along the river branches. From an ecological point of view this results point out the great relevance of the river both like element of continuity and ecological corridor: if riparian vegetation and forests are undamaged the river branches can helps to maintain strictly joined and connected all watershed areas with positive implications for species diffusion, circulation, and reproduction. Finally avoid species isolation river corridors promotes biodiversity maintenance [McArthur and Wilson, 2001].

The IJI (Fig. 6) values stresses that the middle part of the Aniene watershed has got a better land use repartition: all patches are well intermixed and uniformly distributed (salt and pepper mixture). In this area we found a good heterogeneity in patches assemblage furthermore the values calculated at local level with moving window can be localized per area helping to highlights occurring hot-spot in heterogeneity of patches distribution: upstream areas seems to be more homogeneous and less interspersed.
Figure 5 - Connect index values at local scale. Moving Window width 5 Km. Landscape level calculation. Hydrographic network overlayed.

Figure 6 - Interspersion–Juxtaposition index values at local scale. Moving Window width 5 km. Class level calculation. Hydrographic network overlayed.
Conclusions
The study has been developed integrating techniques and disciplines characteristic of engineering and landscape ecology. The main objectives were achieved but the results are related to methods and limited to a general characterization of the area. From the point of view of a “landscape ecologist” the main changing areas have been identified and the general conditions of the river basin characterized, furthermore a “quantitative ecologist” needs more detailed information at ecosystem level: nutrients concentration both in water and riparian soils, biodiversity in detritus decomposers, input and output of organic matter, decomposition rates and metabolic balance of the river. He also needs information and quantitative data about trophic-webs in the water course, diversity of macrophytes and naturally data at community level. We must necessary explain the study results like a starting point to a successive study of characterization of watershed conditions at community and ecosystem level. Landscape ecology and remote sensing could be a useful tools to identify environmental fragility areas and to quantify changes at the opportune temporal and spatial scales, this disciplines can also integrate and supports successfully ecology research but can not replace on field sampling and quantitative ecology studies.

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