Assessing land take by urban development and its impact on carbon storage: Findings from two case studies in Italy

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Abstract

Land take due to urbanization triggers a series of negative environmental impacts with direct effects on life quality for people living in cities. Changes in ecosystem services are associated to land take, among which the immediate C loss due to land use conversion. Land use change monitoring represents the first step to quantify the land take, its drivers and impacts. To this end, we propose an innovative methodology for monitoring land take and its effects on ecosystem services (in particular, C loss) under multi-scale contexts. The devised approach tested in two areas with similar size but different land take levels occurred along the time-span 1990-2008 in Central Italy (the Province of Rome and the Molise Region). The estimates of total coverage of urban settlements at both inventory occasions are provided by point sampling. The area of the urban patches including each sampling point classified as urban settlement at the year 1990 and/or at the year 2008 are used to estimate total abundance and average area of urban settlements. Accounting on carbon loss associated to land take in terms of biophysical and economic values is provided using the InVEST model.

Although land take results 7-8 times higher in the Province of Rome (from 15.1% in 1990 to 20.4% in 2008), our findings show that its relative impact on C storage is higher in Molise, where the urban growth affects consistently not only croplands but even semi-natural land uses such as grasslands and other wooded lands. The total C loss due to land take has been estimated in 1.6 million Mg C, corresponding to almost 355 million €.

Finally, the paper discusses the main characteristics of urban growth and their ecological meaning leading to risks and challenges for future urban planning and land use policies.

Keywords: urbanization, monitoring, sampling approach, ecosystem services, InVEST

Introduction

Urbanization represents one of the main source of disturbance and alteration of natural ecosystems (Churkina, 2008; Imhoff *et al.*, 2004; Solomon *et al.*, 2007), inducing the loss of several ecological functions (Foley *et al.*, 2005). Land take, defined here as the area of land that is converted into settlements and artificial surfaces due to urban growth, alters environmental quality (Ellis and Ramankutty, 2008) and affects the provision of several ecosystem services, such as those related to climate and water regulation (Seto *et al.*, 2012; Nelson *et al.*, 2010). These environmental impacts produce direct and indirect effects on the quality of life of people living in cities (Chiesura, 2004; EEA, 2006; Escobedo *et al.*, 2011; Elmqvist *et al.*, 2013).

Urban areas emit a high proportion of the greenhouse gas carbon dioxide (Svirejeva-Hopkins *et al.*, 2004) and contribute somewhere between 40 and 85% of total anthropogenic greenhouse-gases (GHG) emissions (Satterthwaite, 2008). The effects of urbanization on climate change are exacerbated by the loss of carbon (C) pools associated with the decreases in the vegetative cover caused by the land take associated to the expansion and intensification of urban areas (Hutyra *et al.*, 2011a). Moreover, soils in urban areas have very low C densities (Pouyat *et al.*, 2006), producing negative impacts of urbanization on C sequestration. Land take by urban development determines a loss in the carbon stock, as well as in the future carbon uptake potential by the land (Hutyra *et al.*, 2011b). A few studies addressed this problem, by proposing methodologies to assess the carbon impact of growing urban regions. Seto *et al.* (2012) modeled the loss in aboveground biomass carbon from areas with high probability of urban expansion until 2030, and concluded that this loss is likely to be significant (equal to ~5% of emissions from tropical deforestation and land-use change). Raciti *et al.* (2012) focused on the effects of urbanization on soil carbon pools, by comparing the carbon content of open areas and impervious-covered soils. Their finding is that carbon content under impervious surfaces is 66% lower. Hutyra *et al.* (2011a) estimated the carbon consequences associated to urban land take in the Seattle metropolitan region, and concluded that it represents a substantial term in the regional carbon balance.

Despite all these studies suggest that the loss of carbon stock (and future carbon uptake) due to land take by urban development is potentially significant, this effect is often overlooked during the assessment of the future impacts of urban growth. For example, the treatment of climate-related issues in Strategic Environmental Assessment (SEA) of spatial and urban planning is still quite weak and largely based on general recommendations, as opposite to analytical evaluations (Geneletti, 2015). There is a need for further development of methods to assess the impact of land take on carbon storage, that can be transferred to practitioners and used to support the proposal of more sustainable urban plans and policies. Particularly, these methods need to address two issues: the analysis of land take dynamics, and the modeling of carbon loss associated to them.

The objective of this paper is to contribute to fill this gap by proposing and testing a method to quantify land take dynamics associated to urban growth, and estimate their effects in terms of carbon storage loss. Land take dynamics were analysed through the construction of transition matrices (Pontius *et al.*, 2004; ONCS, 2009). Particularly, a method proposed by Baffetta *et al*. (2011) and used for urban forest coverage assessment over Italy (Corona *et al.*, 2012a) was implemented in order to estimate abundance and average size of the urban patches. Subsequently, the sampled urban patches were used as input for the assessment of change in carbon loss, both in biophysical and economic terms.

The study areas are represented by the Province of Rome and the Molise Region, in Italy (see Figure 1). These two areas were selected because of their different socio-economic context leading to different population density and urban growth patterns. This produced a typical polycentric urban form in Rome, and a very fragmented urban growth in Molise characterized by small patches mostly surrounded by rural lands. In Italy urban areas cover 7.1% of the land area, and grew by 500,000 in the period 1990-2008, at the expense of croplands in plains and low hills (Corona *et al.*, 2012b; Marchetti *et al.*, 2012a). However, few studies addressed urban growth and its impact in Italy (Romano and Zullo, 2013), due to the lack of reliable information and the high costs of updating. This lack highlights the need of improve land use monitoring systems and the need to develop new methodologies aiming to increase their informative power while containing the costs of realization and updating.

Materials and methods

Study area and available data

Analysis are carried on two very different study areas in central Italy, one of the ancient human dominated areas within the Mediterranean Basin, which has been indicated by Myers *et al.* (2000) as one of the four most significantly altered hotspots on Earth (Fig. 1). In these areas natural capital has been altered by human population for thousands of years (Falcucci *et al.*, 2007) and its pressure is still rising, especially along the coast (Salvati *et al.*, 2012; Romano and Zullo, 2014). The Province of Rome is one of the most populated and urbanized areas in Italy. It covers about 5,352 km^2 with a total population of 4,061,543 inhabitants (ISTAT, 2008). The territory mainly consists of hills (50%) and lowlands (30%), while mountains cover only a minor proportion (20%). Rome followed the spatial pattern of Mediterranean cities with a rapid transition from the traditional compact model to a scattered and polycentric urban form, characterized by huge expansion around the urban area (Salvati, 2013).

On the contrary, the Molise region is among the least dense and urbanized area in Italy, with a negative demographic rate during the past decades (ISPRA, 2014a; Sallustio *et al.*, 2013). This region has an extension of 4,438 km^2 with a total population of 313,660 inhabitants (ISTAT, 2008); mountainous (55.3%) and hilly (44.7%) areas cover the whole territory. Fig. 2 shows land use of the two study areas at the year 2008.

Currently, different land use and land cover maps are available for both of the study areas. They are usually achieved through digital mapping processes or combination of automatic and semiautomatic processes of classification. The former is the case, for example, of Corine Land Cover (Maricchiolo *et al.*, 2004) and land use maps produced by the regional administrations, obtained through the visual interpretation of high-resolution ortophotos or satellite images. The latter is the case of the high resolution layers made available by the GMES Copernicus program (EEA, 2013). Despite their recognized utility, the scarce possibilities of updating, usually due to the high costs of images and photo-interpretation processes, represent serious barrages on using land use maps for monitoring land use and land cover change (LULCC) through time.

To overcome these limits, several inventory approaches have been developed and applied as a reliable alternative for LULCC monitoring. In Italy, different project are focused on LULCC using an inventory approach such as: the National Land Take Monitoring Network, performed by ISPRA using a stratified sampling methodology, which combines orthophotos interpretation and remote sensing data with high-resolution (ISPRA, 2014a); the AGRIT project, where the sampling is based on an area frame from 1988 to 2000 and where a point frame (project POPOLUS) was introduced in 2001 (MIPAF, 2014)*;* the Land Use Inventory (IUTI), based on point sampling and implemented by the Italian Ministry of Environment, Land and Sea as an instrument of the National Registry for forest carbon sinks for accounting GHG emissions (Corona *et al.*, 2012b).

Land take assessment

Classification method and urban patches delineation

The IUTI dataset was used in this study due to specific characteristics: large sample size, easy upgrades and uncertainty value estimate (Corona *et al.* 2007; Corona, 2010).The IUTI approach is here tested in order to estimate urban growth occurred from 1990 to 2008, and furtherly developed to estimate the changes in urban patches abundance and their average area at the two inventory occasions.

Localization of sampling points was carried out according to a tessellation stratified sampling design (also known as unaligned systematic sampling; Barabesi and Franceschi, 2011). The set of sample points was extracted using a 0.5 km square grid, geo-referenced and randomly located in each square cell and fully covering the study area. A total of 21,412 sample points were extracted for the Province of Rome and 17,737 for the Molise Region.

Each sample point was photo-interpreted and classified according to the IUTI classification in Table 1 (for details, see Corona *et al.*, 2012b). It was carried out a visual interpretation and diachronic analysis of digital aerial orthophotos acquired at the years 1990 and 2008: 1990, TerraItaly 1988/1989, panchromatic aerial orthophotos, with spatial resolution of 1 m; 2008, TerraItaly 2008 dataset, digital color aerial orthophotos with spatial resolution of 0.5 m. For each sampling point, classified as urban at the year 1990 and/or at the year 2008, the urban patches including the sample point were mapped for both inventory occasions.

An overlap analysis was performed in order to identify patches transformed from other land use classes to urban, during the considered time-span. The previous dominant land use class (at the year 1990) was also assigned according to the IUTI classification to each new urban patch at the year 2008. The outputs of this diachronic analysis were the land use classification for each sample point at the years 1990 and 2008 and the corresponding map of urban patches (Fig. 3) and their classification.

Estimation of coverage, abundance and average size of urban settlements

Let A, N and $\bar{a}$ be, respectively, the coverage, abundance and average size of the urban settlements in each study case, and be Q the extent of the area formed by the n squares completely covering the investigated territory under the tessellation stratified sampling scheme adopted by IUTI.

According to Corona *et al.* (2012a), the estimate of A is given by

$$\hat{A} = \hat{p}Q$$

where:

$$\hat{p} = \frac{n_u}{n}$$

n_u = number of sample points classified as urban settlements.

The variance of $\hat{A}$ can be estimated as

$$\hat{\mathrm{var}}(\hat{A}) = Q^2 \frac{n_u(n-n_u)}{n^2(n-1)}$$

Let S and a_j be, respectively, the set of urban patches selected at least once by the n sampling points and the size of the j-th urban patch. According to Corona *et al.* (2012a), if the a_js are negligible with respect to Q the estimate of N is given by

$$\hat{N} = \frac{Q}{n} \sum_{j \in S} \frac{1}{a_j}$$

with estimated variance equal to

$$\hat{\mathrm{var}}(\hat{N}) = \frac{1}{n(n-1)} \left(Q^2 \sum_{j \in S} \frac{1}{a_j^2} - n\hat{N}^2 \right)$$

Accordingly, the estimate of $\bar{a}$ is given by $\hat{A}/\hat{N}$, i.e.

$$\hat{\bar{a}} = \frac{n_u}{\sum_{j \in S} \frac{1}{a_j}}$$

with estimated variance equal to

$$\hat{\mathrm{var}}(\hat{\bar{a}}) = \frac{Q^2}{\hat{N}^2 n(n-1)} \sum_{j \in S} \left(1 - \frac{\hat{\bar{a}}}{a_j} \right)^2$$

Carbon loss assessment

Changes in C storage due to the urbanization is based on differences of C stocked by different land use classes, and its loss or gain related to the transition from one class to another through time. Several ecosystem mapping and assessment tools with different aims and characteristics have been published during the last years, such as ARIES (Villa *et al.*, 2014), InVEST (Nelson *et al.*, 2013; Nelson and Daily, 2010) EcoAIM, ESR, ESvalue, NAIS, EcoMetrix, etc. (Martínez-Harms and Balvanera, 2012; Waage *et al.*, 2011). We decided to assess the changes of C stock both in biophysical and economic terms using the InVEST (Integrated Valuation of Environmental Services and Tradeoffs) Carbon Storage and Sequestration model developed by the Natural Capital Project team (Daily *et al.*, 2009; Tallis et al., 2013). We decided to use InVEST for different reasons, among which: a) it is a free and open-source software; b) it is organized in different tiers of difficulty of use and input data availability; c) it is able to assess several ecosystem services; d) it is based on the application of the production function approach, able to provide more accurate and policy-relevant results (Nelson and Daily, 2010). These characteristics enable its use in different contexts and its further future implementation for mapping and assessment of other ecosystem services and their trade-offs. InVEST is a geospatial modeling framework tool that predicts the provision and value of ecosystem services, using land use/land cover maps and related biophysical, economic and institutional data. InVEST employs a simplified carbon cycle and evaluates the impact of LULCC on ecosystem services (Nelson *et al.*, 2009; Polasky *et al.*, 2011).

The model works applying the estimates of carbon stored by each land use (LU) class to produce a map of carbon storage for the considered carbon pools. For each class, the model requires an estimate of the amount of carbon stored by each of four fundamental C pools.

All the data concerning carbon storage were assigned according to the Good Practices Guidance for Land Use, Land Use Change and Forestry (GPG-LULUCF) classification and definition: living biomass (above ground and below ground), dead organic matter (dead wood and litter) and soil (soil organic matter in the upper 30 cm) (Woomer *et al.*, 2004; Gockowski and Sonwa, 2011; Adu-Bredu *et al.*, 2011; Asase *et al.*, 2011; Yao *et al.*, 2010; Leh *et al.*, 2013).

Values for each C pool and LU class were attributed applying the IPCC methodology (IPCC, 2003, 2006), using data from bibliography review (Table. 2). We assumed, according to the conservative approach proposed by the tier 1 of the Guidelines for National Greenhouse Gas Inventories (IPCC, 2006), that settlements do not contribute to C storage. Moreover we did not take into account C stored in wetlands because land take did not occurred at the expense of this LU class during the observed time-span.

For the C storage at the year 1990 we assumed the carbon storage equilibrium (steady-state level) due to its long persistence in each grid cell. Therefore, this dataset was implemented in all the mapped patches to assess the net change in C stock due to land take in the period 1990-2008, assuming that change in carbon stocks is only due to LULCC.

Here we report the economic value of carbon storage as the Social Cost of Carbon (SCC). The SCC, also known as the marginal damage cost of carbon dioxide, is defined as the net present value of the incremental damage due to a small increase in carbon dioxide emissions. The choice of SCC is due to the fact that it would be equal to the Pigouvian tax that could be placed on C (Tol, 2009). Therefore, in our case, the final total SCC hypothetically represents the social cost related to the land take over time. We used a SCC of 37$ Mg^{-1} of CO_2 (about 109€ Mg^{-1} of elemental C) (OIRA, 2013), and assumed this value stable from 1990 to 2008. We used a discount rate of 7% per year, which is one of the typical values suggested used for cost-benefits analysis of environmental projects (Stern, 2007). The discount rate has been used to refer all the economic values to a unique time, the 2008. Moreover, this value falls within the range of 5-10% per annum suggested by several economic studies (e.g. Nordhaus, 2007), avoiding the misallocation of monetary resources and judge climate change mitigation activities similarly to all other policies. Thinking about urban policy and planning, this estimate could represent an attempt to internalize the externality and restore the market to the efficient solution. The results in terms of biophysical and economic benefits obtained for the sample patches were extended to the whole surface by statistical inference.

Results

Estimation of urban area

Table 3 shows the main results of the statistical survey carried out. Sampling points classification and polygon delineation at the year 1990 and 2008 highlight that urban areas in the Province of Rome covered about 15.1% of the whole territory at the year 1990 and has increased to 20.4% at the year 2008. Urban areas in Molise are smaller, increasing from to 2.0% of the land area to 2.9% during the period 1990-2008 (Fig. 4). Land take by urban areas expansion amounted to 28,000 ha and 3,850 ha in Rome and Molise, respectively. The Province of Rome showed a smaller increase in total urban area compared to Molise, with a percentage increase of, respectively, 35% and 45% with respect to the 1990.

Besides urban densification within the cities' cores, urban sprawl continue to affect the rural landscape under a contagion (*sensu* Ricotta et al., 2003) pattern, as demonstrated by the increase of urban patches number and the substantial stability of their size. This phenomenon has an important meaning on economic, social and environmental impact of urban growth. It is particularly evident in Molise, where the urban patches number increased by 41% at 2008 and the increase in the average urban patch size is less than 1%, respect to the 3.4% of Rome.

The estimated average area of urban patches has remained stable from 1990 to 2008, with a slight increase in both territories. The average area of the urban patches is twice in Rome. The number of estimated urban patches is higher in Rome, but the increase observed during the considered time-span is lower compared to Molise. In fact, the number of urban patches has increased of 41% in Molise with respect to 30% estimated for Rome. The Molise region shows higher standard errors for all the estimators compared to Rome, depending on the less consistency of urban areas.

The land take occurred at the expense of different LU classes in the two study areas (Fig. 5). Although in both cases the majority of the phenomenon has affected on croplands, this is particularly evident in Rome, where about the 90% of the land take occurred on this class. This percentage is lower in Molise (about 61%), where land take is also remarkable on pastures and grasslands.

Carbon loss between the years 1990 and 2008

Average carbon densities in terrestrial LU decrease with increasing moving from natural to human-influenced LU classes. We found the highest C densities in forests (143.42 Mg C ha^{-1}), while the lowest is in the croplands (58.1 Mg C ha^{-1}). The smallest C density's values were found in dead organic matter and below ground biomass, while the greatest is stored in soil.

Change in C storage due to land take on the sample patches estimated using the InVEST model proves that: in Molise Region carbon stock has decreased about 111,954 Mg, while in the Province of Rome carbon loss is about 904,536 Mg.

By statistical inference these results of carbon losses can be extended to the whole area so that from 1990 to 2008 the Molise region had a total decrease of 252,335 Mg C (about 14,018 Mg C year^{-1}), corresponding to a total economic loss at the year 2008 equal to 54,619,258 € (about 3,034,403 € year^{-1}). The same estimates for Rome's territory provide a total loss of 1,390,234 Mg C (about 77,235 Mg C year^{-1}) and an economic loss at the year 2008 equal to 300,922,997 € (about 16,717,944 € year^{-1}). This mean that C loss due to urbanization occurred between 1990 and 2008 amount to 59.7 and 65.5 Mg C ha^{1} in Rome and Molise, respectively. This corresponds to a mean economic loss of 12,920 € ha^{-1} in Rome and 14,186 € ha^{-1} in Molise.

These values represent the average SCC per hectare related to the urbanization. The higher values estimated in Molise (both in biophysical and economic terms) are related to the high percentage of land take occurred at the expense of LU classes such as grasslands, which have a higher total C density than croplands.

Discussion

Ecological meaning of land take

The study of the land take in the two different areas highlighted the higher severity of the urban growth issue in the Province of Rome with respect to the Molise Region. Indeed, while at 2008 settlements covered about 19.6% of the Rome's territory, gaining 4.9% respect to 1990, it represented only the 2.9% of the Molise's territory, gaining 0.9% respect to the baseline.

This dramatic increase in settlements in both study areas led to a decrease in C stocks. Although the total C loss is higher in Rome, the unitary values of this loss are higher in Molise (+9.7%, corresponding to 5.8 Mg C ha^{-1}), due to the higher incidence of land take on semi-natural land uses which are characterized by relatively high values of C densities. In accordance with other studies, the ecological consequences of urban growth are strongly related to the previous land use (Jenerette *et al.*, 2006; Pouyat *et al.*, 2006; Pickett *et al.*, 2008). We can conclude that there is a sort of anthropogenic gradient affecting the urbanization impact on C loss. Therefore, the higher the naturalness of the territory, the higher the C loss. This is the case of forests, which, as even demonstrated by Zhang *et al.* (2012), once converted are responsible of the highest unit values of C loss (Fig. 6).

To give a sense of the magnitude of the C loss associated to land take in the study areas, we compared it with the C stock and sink in the National Park of Abruzzo, Lazio and Molise (PNALM), one of the largest National Park in Italy, which includes parts of the two study areas. The forest cover in the PNALM is about 37,962 ha, mainly dominated by beech forests. C stocked by forest aboveground biomass is about 3.3 M Mg C and the C sink is about 65,900 Mg C $year^{-1}$ (Marchetti *et al.*, 2012b). This means that the total C loss due to the urban growth occurred between 1990 and 2008 in Molise and Rome corresponds to about 49.5% of the C stocked by forests within the PNALM. Concerning the annual C loss, urbanization in the two study areas amounts to 91,253 Mg C $year^{-1}$, exceeding then the forest annual C sink of the PNALM for the 38.5%.

Yet, in order to increase the awareness of policy makers on the C footprint of urban growth, it could be interesting to simulate the implementation of project such as the realization and maintenance of urban green spaces as a mitigation strategy. Strohbach *et al.* (2012a) estimated a C sequestration between 137 and 162 Mg CO_2 ha^{-1} (37.3 and 44.1 Mg C ha^{-1} respectively) for a 50 years urban green space project in Germany. In our case, this would correspond to the realization of 40,000 hectares of urban green spaces to balance the C loss related to the total urban growth occurred in both the study areas. This represents a topical figure that is almost coincident with the total amount of the urban forests coverage currently present in the whole Italian territory (43,000 ha; Corona *et al.*, 2012a)

Risks and opportunities towards new paradigms for urban planning

LULCC is not always related to population growth, and other individual and social conditions must be taken into account (Lambin et al., 2001). This is particularly evident in our case studies, where the increase in the number of urban patches and the stability of their size, especially in the Molise Region, combined with the densification of urban fabric within the cities, highlighted the duplicity of the urban growth patterns: compact but often under-used inside existing cities boundaries (urban shrinkage), more fragmented and scattered outside them (urban sprawl). With particular regard to the latter, as reported by Romano and Zullo (2012), in Italy, during the past 50 years, the urban growth has been characterized by "a huge dispersion in diffused forms scarcely governed by interpretable rules, leading to the systematic reproduction of a city model lacking town planning". Moreover, different urban forms and spatial structures may imply different forms of land take containment, especially in terms of green infrastructure planning in the vulnerable areas as wildland-urban interface (WUI; Elia *et al.* 2014). The quantitative limits to the admissible increase in urbanized areas are not sufficient. Additional parameters should be considered, among which the shape of such areas, the territorial dispersion indices and the density and types of transport networks (Romano and Zullo, 2013).

On the other hand, analyzing demographic data provided by the National Institute of Statistics, between 1990 and 2008 the Province of Rome grew by about 110,000 inhabitants (+2.9% respect to 1990), while the trend has been negative in Molise (-13,000 inhabitants, corresponding to about 4% of the population at 1990). In our case it is particularly evident that the urban growth is completely independent from housing demand related to the demographic trend. The consequent decrease in population within cities leads to the concept of shrinking cities (Haase *et al.*, 2014), which is considered an important issue especially in Europe (Turok and Mykhnenko, 2007; Kabisch and Haase, 2011). Urban shrinkage involves at the same time new risks, challenges and opportunities for future land use policy and management. It implies dramatic impacts such as: under-utilization, de-densification and vacancy, demolition and resulting gray fields and brownfields in the compact areas (Schilling and Logan, 2008), new land take outside (Salvati *et al.*, 2012) and other problems related to the policy and management strategies (Couch *et al.*, 2012): furthermore, as partially demonstrated in our study, the removal of vegetation, the addition of impervious surfaces and increases in local fossil fuel usage due to urban growth, are typically associated with significant carbon emissions (Hutyra, 2011a). On the other side, the effect of urban shrinkage on urban spaces, especially those related to de-densification and vacancy, offers great potential to "re-create", enhance and implement urban green space (Haase *et al.*, 2014). The implementation of new green spaces and green infrastructure in urban and peri-urban areas may lead to the enhancement of several ecosystem services provision, among which C storage and sequestration (Strohbach *et al.*, 2012a).

The assessment of biophysical and economic consequences of land use changes on ecosystem services may represent an important tool for public administrations (Cimini *et al*., 2013; Marchetti *et al*., 2014), which can be applied during the SEA of their policies, plans and programs. For example, the SCC may be used to suggest suitable form of taxation on urban growth, as a way to compensate its negative effects. This hypothetical Pigouvian tax may have a double effect: a) disincentive land take promoting the requalification of existing settlements; b) offer the possibility to the administrations to invest the income in urban green space projects to mitigate the negative effects of new urbanization. In this perspective, our estimates of 12,920 € ha^{-1} in the Province of

Rome and 14,186 € ha^{-1} in Molise region may represent a first attempt to define appropriate off-set measures for land take. However, it is important to remember that in our case these values refer only to carbon storage, ignoring all the other ecosystem services and their positive or negative trade-offs. An additional result of this economic tool, consisting to relieve the pressure of urban growth on land uses with high ecosystem services value (e.g. forests), could be achievable diversifying the value of the tax depending on the previous land use. In our case, for example, this distinction would result about 31,043 € ha^{-1} in the case of urbanization at the expense of forests or 12,576 € ha^{-1} in the case of urbanization at the expense of croplands (Fig. 6).

Considering that approximately 78% of the European population lives in urban areas (EEA, 2006), it is fundamental to include consideration of ecosystem services during impact assessment of urban policies and plans, in order to promote urban sustainability and resilience. Ecosystem services assessment and valuation represent a potentially helpful support tool, especially for awareness raising, economic accounting, priority-setting, incentive design, alternative comparison and conflict resolution (TEEB, 2010; Barton et al., 2012; Gómez-Baggethun and Barton, 2013; Geneletti, 2013).

Limitations of the study

Although several studies demonstrate that urban areas are able to store reasonable quantities of C (Larondelle and Haase, 2013; Zhang *et al.*, 2012) in green spaces and urban trees (e.g. Tao *et al.*, 2014; Vaccari *et al.*, 2013; Strohbach *et al.*, 2012b; Churkina *et al.*, 2010; Tian *et al.*, 2008; Pickett *et al.*, 2008; Han *et al.*, 2007) we decided to not consider these quantities. In accordance with the IPCC- tier 1 guidelines (2006), we opted for a more conservative approach, which gave us the opportunity to set up a sort of carbon baseline related to urban growth. Despite of this limit, mainly related to the underestimation of C stock in urban areas, such a conservative approach proves to be particularly useful in our case studies for different reasons, among which: a) the absence of inventory data on C storage in urban green spaces (urban forests, garden, boulevard etc.) applicable at regional scale; b) the huge variability of C stored by urban green spaces, which is heavily affected by the age of the stands, their design and management (Strohbach *et al.*, 2012a), suggesting to avoid the use of data from literature; c) the conservative approach could be particularly suitable to increase the awareness of policy makers in context like the Italian one, where, despite the international commitment to reach the no (zero) net land uptake by 2050 (European Commission, 2011), the actual trend results far from this objective. Condtions in many countries are characterized by quite similar limitations, and thos stresses the portability of the proposed conservative approach. Furthermore, this underestimation is partly balanced by the use of 0.5 ha and 20 m width as minimum thresholds for the identification of urban areas, thus excluding the isolated houses and the scarcely dense settlements, which are well represented especially in rural contexts, and their impact on C storage.

Conclusions

The fast growth of the urbanization, especially in sensitive areas represents a global issue. Monitoring urban growth is crucial, especially in Europe, where the target is to achieve the objective of no (zero) net land uptake by 2050 (European Commission, 2011).

In this paper we tested an innovative methodology for monitoring land take and its effects on ecosystem services (in particular C loss) widely applicable to other multi-scale contexts. Such a methodology could be particularly helpful in contexts where there is a lack in coordinated survey activities and LULCC monitoring. This is the case of Italy, where local and regional studies and monitoring programs use different methodologies usually based on wall to wall land use mapping lacking on statistical accuracy. Differences in methodologies, time-span coverage and land use classification systems used among different territorial contexts, lead to the difficulty in a) data standardization; b) their use to support SEA of land use policy and planning at larger scale; c) the comparison across different territorial contexts. The latter, particularly, could be a helpful approach to control and verify the effect of different land management strategies promoting the implementation of best practices.

This study intended to present an approach easily applicable at different spatial scale even if they are characterized by the lack of available input data. Despite the low realization and updating costs, the integration of inventory and cartographic approach proves suitable for providing reliable estimates, enhancing at the same time their information potential. Moreover, the possibility to couple such estimates with a spatial explicit tool like InVEST allows to identify hotspot of C loss due to urban growth, thus providing a useful support for land use planning.

The accounting of the economic value related to C loss could act as an additional tool able to enhance the awareness of policy makers on urban growth impact on ecosystem services. Impacts on some ecosystem services are often neglected during SEA of land use planning because of their economic invisibility with respect to other issues, such as urban infrastructure, settlements, etc. The assessment of impacts on ecosystem services, mainly in human dominated ecosystems, may help to reconcile the historical bias between nature and human improving and completing the costs-benefits analysis related to particular choices, policies, plans and projects (Jansson, 2013). Therefore, it will play an important role supporting future policies aimed to satisfy human needs but at a smaller cost on natural systems (Millennium Ecosystem Assessment, 2005).

Our results strongly encourage the joint use of monitoring approaches of land take with ecosystem services assessment and valuation to better understand the concept of sustainability in urban areas and its implications on other ecosystems. As suggested by Larondelle and Haase (2013), to use and maintain resources sustainably, land use decision-making processes have to incorporate ecological principles considering an a urban-rural continuum.

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Fig.1: Geographical location of the study areas

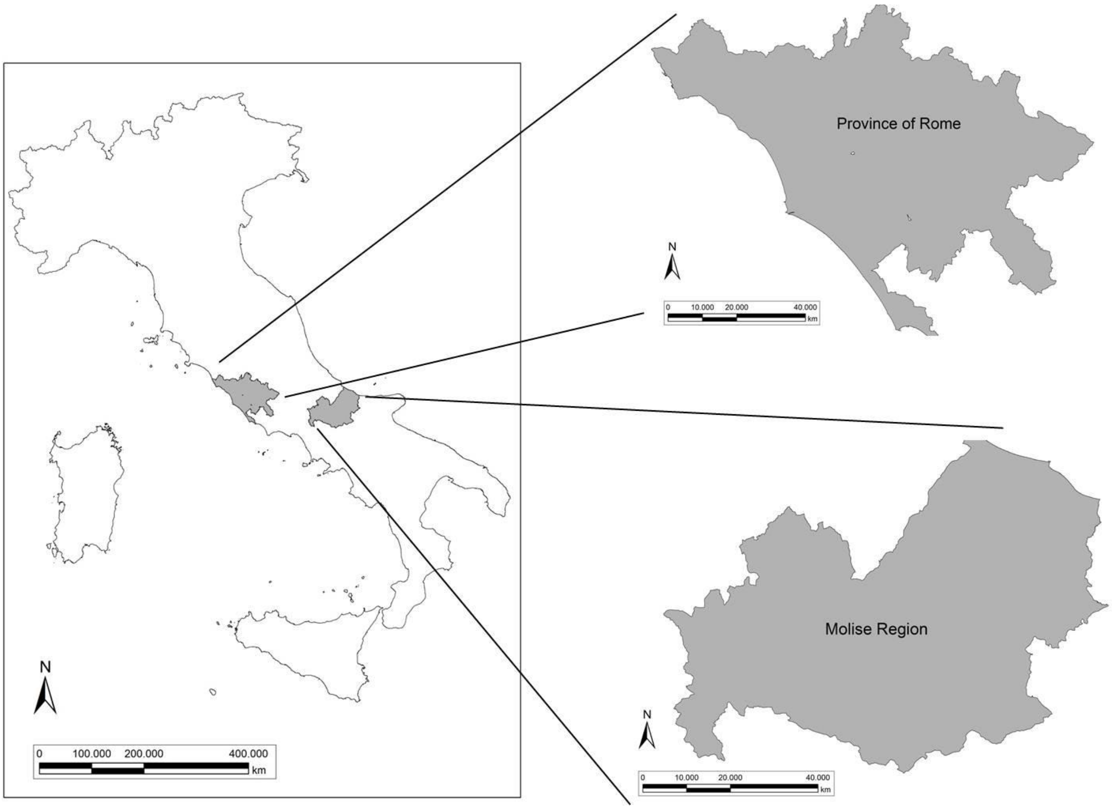
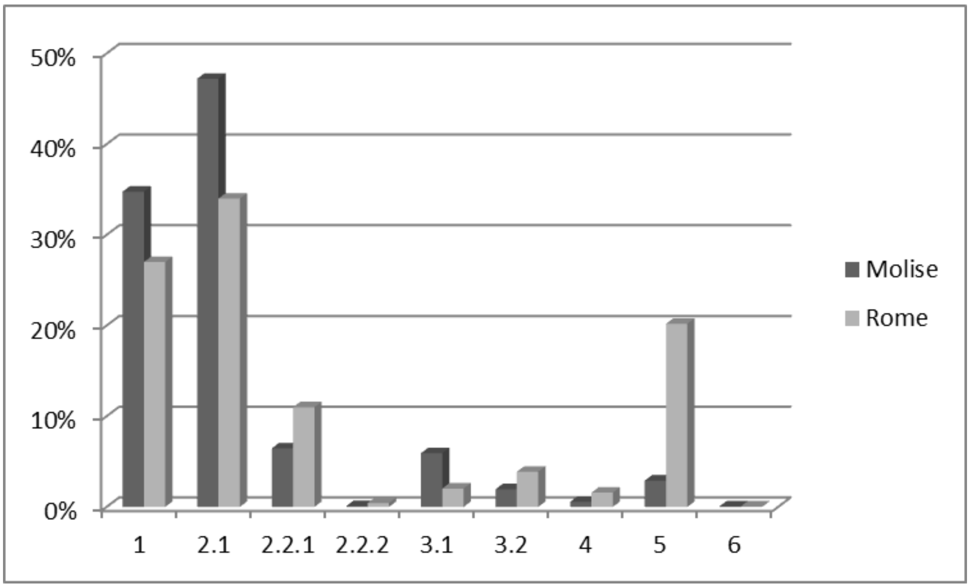


Fig.2: Land use classes in the Molise Region and in the Province of Rome at the year 2008, according to IUTI classification (Tab. 1).



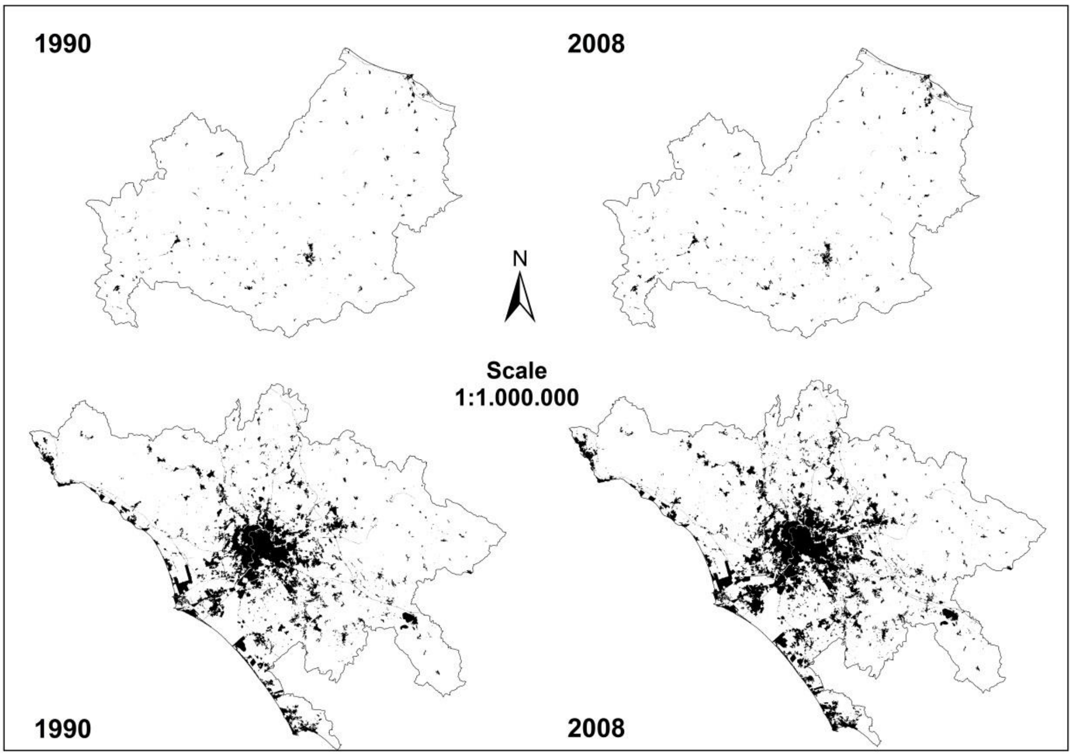
Tab. 1 - IUTI land use classification (Corona et al., 2012b)

IUTI class	*IUTI category/subcategory*		*IUTI code*
Forest land	-		1
Cropland	Arable land		2.1
	Permanent crops	Orchards, vineyards and nurseries	2.2.1
		Forest plantations	2.2.2
Grassland	Natural grassland and pastures		3.1
	Other wooded land		3.2
Wetlands	-		4
Settlements	-		5
Other land	Bare rock and sparsely vegetated areas		6

Tab. 2-C stocks (Mg C ha^{-1}) in each terrestrial LU class and C pool and references used for the C pools' values. (*) According to the tier 1 proposed by IPCC (2006), the most conservative approach has been used, meaning that urbanization causes carbon stocks to be entirely depleted. (**) Concerning other land converted to settlements, change in carbon stocks has been not estimated, according to the GPG (IPCC, 2003), as no change in carbon stocks in the other land has been assumed.

IUTI code	*Above ground (Mg C ha^{-1})*	*Below ground (Mg C ha^{-1})*	*Dead Organic Matter (Mg C ha^{-1})*	*Soil Organic Carbon (Mg C ha-1)*	*Total (Mg C ha^{-1})*
1	50.5 (Gasparini & Tabacchi, 2011)	11.525 (Est. ISPRA, 2014b)	5.295 (Gasparini & Tabacchi, 2011)	76.1 (Gasparini & Tabacchi, 2011)	143.42
2.1	5 (ISPRA, 2014b)	/	/	53.1 (Chiti *et al.*, 2012)	58.1
2.2.1	10 (ISPRA, 2014b)	/	/	52.1 (Chiti *et al.*, 2012)	62.1
2.2.2	28.55 (Gasparini & Tabacchi, 2011)	5.25 (Est. ISPRA, 2014b)	1.75 (Gasparini & Tabacchi, 2011)	63.9 (Gasparini & Tabacchi, 2011)	99.45
3.1	/	/	/	78.9 (ISPRA, 2014b)	78.9
3.2	3.05 (IPCC, 2003)	/	/	66.9 (ISPRA, 2014b; Alberti *et al.*2011)	69.95
5	*	*	*	*	*
6	**	**	**	**	**

Fig. 3: Urban patches at the years 1990 and 2008 in Molise Region (above) and in the Province of Rome (below).



Tab. 3 - Estimates of number of urban patches ($\hat{N}$), urban coverage ($\hat{A}$) and urban patch average area ($\hat{\bar{a}}$), and thei restimated relative standard errors (expressed in percent)

Study area	*Year*	$\hat{N}$	$se_{\hat{N}}$ (%)	$\hat{A}$ (ha)	$se_{\hat{A}}$ (%)	$\hat{\bar{a}}$ (ha)	$se_{\hat{\bar{a}}}$ (%)
Molise Region	1990	2,449	10.1	9,000	5.2	3.68	7.6
	2008	3,455	8.5	12,850	4.3	3.72	6.2
Province of Rome	1990	10,763	4.9	81,037	1.7	7.53	4.4
	2008	13,989	4.2	109,026	1.4	7.79	3.8

Fig. 4: Urban coverage in1990 and land take occurred between 1990 and 2008 in the study areas

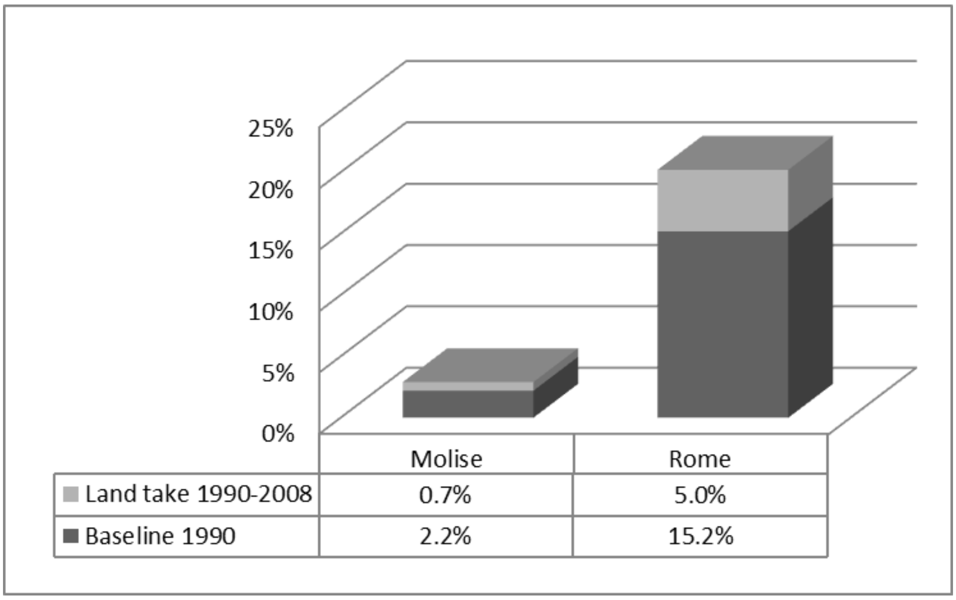


Fig. 5: LU classes affected by land take between 1990 and 2008 in the study areas.

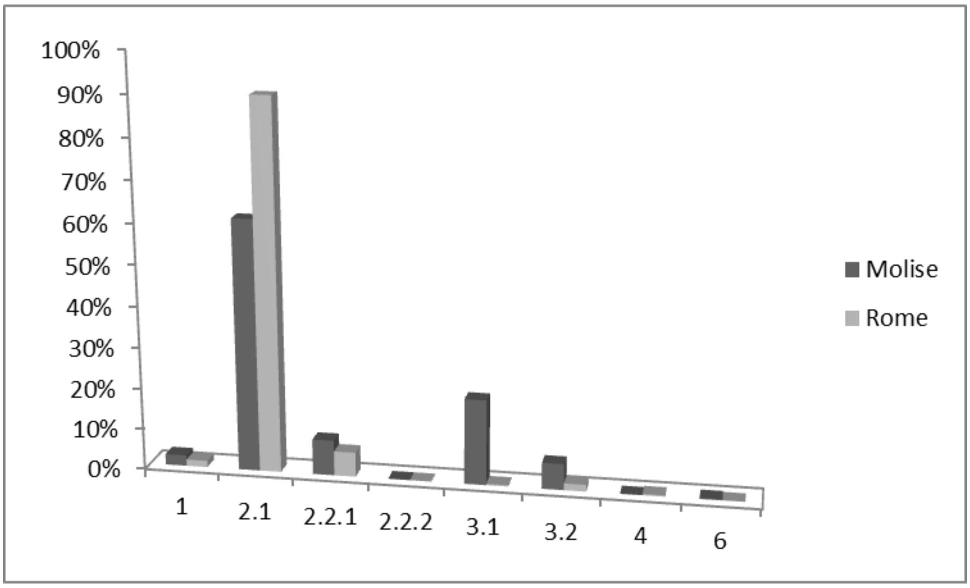


Fig. 6: Urban growth at the expense of different LU classes involves different impacts on C loss, both in biophysical and economic terms (Detail of the study area)