nature sustainability

ANALYSIS
https://doi.org/10.1038/s41893-019-0340-0

# Direct and indirect loss of natural area from urban expansion

Jasper van Vliet

**Global losses of natural area are primarily attributed to cropland expansion, whereas the role of urban expansion is considered minor. However, urban expansion can induce cropland displacement, potentially leading to a loss of forest elsewhere. The extent of this effect is unknown. This study shows that indirect forest losses, through cropland displacement, far exceed direct losses from urban expansion. On a global scale, urban land increased from 33.2 to 71.3 million hectares (Mha) between 1992 and 2015, leading to a direct loss of 3.3 Mha of forest and an indirect loss of 17.8 to 32.4 Mha. In addition, this urban expansion led to a direct loss of 4.6 Mha of shrubland and an indirect loss of 7.0 to 17.4 Mha. Guiding urban development towards more sustainable trajectories can thus help preserve forest and other natural area at a global scale.**

The global competition for land for multiple uses has led to a dramatic loss of natural area[1]. This loss of natural area has had large and negative impacts on terrestrial biodiversity[2] and ecosystem services[3], and the loss of forests in particular has also greatly contributed to global greenhouse gas emissions[4]. Cropland expansion has been identified as the most important proximate cause of the loss of natural area[5,6]. As a consequence, cropland intensification and dietary changes have received much attention as potential solutions to reduce the decline of natural area[7–9].

Contrary to cropland expansion, urban development is only associated with a small fraction of all forest losses[5,6]. Although the relation between urbanization and forest loss has been firmly established, the underlying mechanism that relates urbanization to forest loss is not clear[10]. Land use displacement (that is, the geographic displacement of land use activities[11]) may potentially explain this relation. The conversion of cropland into urban land and the development of new cropland elsewhere to compensate for the loss in production may be interpreted as land use displacement. As urban expansion often takes place in cropland areas[12], and as cropland expansion often leads to a conversion of natural area[6,13], cropland displacement may relate urban expansion to losses of natural area elsewhere. Future land use change scenarios have projected this effect at both local[14] and global[15] scales, but to date there has been no analysis of observed changes.

This paper analyses to what extent urban expansion has contributed both directly and indirectly to the loss of natural area between 1992 and 2015. Direct changes refer to natural area that converted into urban land, whereas indirect changes refer to natural area that converted into cropland to compensate for cropland that was converted into urban land elsewhere; that is, indirect changes are a consequence of cropland displacement. The natural area considered in this study includes forest and shrubland, but excludes grassland because it was not possible to differentiate between managed grassland and natural grassland. The analyses test the hypothesis that indirect losses of natural area exceed direct losses. The analyses also test the hypothesis that differences in cropland productivity leverage cropland displacement; that is, the area of cropland that is required to compensate for the loss in crop production is larger than the area of cropland that is converted into urban land. Both hypotheses build on the observation that urban areas are typically located in highly productive cropland areas[12], whereas new cropland mainly comes at the cost of forest and other natural area[13].

## Results

**Direct land cover changes.** According to the European Space Agency's Climate Change Initiative (ESA CCI) land cover data[16], 38.0 Mha of new urban land appeared globally between 1992 and 2015 (Fig. 1c), representing a 115% increase in only 23 years. About 64% of this urban expansion took place on former cropland, while 9%, 13% and 10% led to a direct loss of forest, shrubland and grassland, respectively (Fig. 1a and Supplementary Table 1). The remaining 5% led to a conversion of other land (mainly bare land). Yet, large differences existed between world regions. For example, more than 75% of the urban expansion in Southeast Asia, India, China and Europe took place on former cropland, whereas urban expansion into former cropland was 40% or less in Oceania, Sub-Saharan Africa, and the Middle East and Northern Africa (MENA). Consistently, in regions where most urban expansion took place on cropland, little urban expansion took place on forest and shrubland, and vice versa (Fig. 1a and Supplementary Table 1).

An analysis of the same data also showed that new cropland mostly led to a conversion of forest (56%) and shrubland (30%), whereas 11% and 3% led to a conversion of grassland and other land, respectively (Fig. 1b and Supplementary Table 2). At a regional scale, the data indicated that large differences existed between land cover types that converted to cropland, mainly related to the prevailing natural vegetation in different regions. For example, cropland expansion in Southeast Asia and Latin America mainly led to a loss of forest, but cropland expansion in Oceania and MENA mainly led to a loss of shrubland. In some regions, notably China, Russia, Central Asia and Sub-Saharan Africa, there was a considerable amount of grassland that converted into cropland.

**Cropland displacement and indirect land cover change.** Urban expansion between 1992 and 2015 led to a direct loss of 3.3 Mha of forest and 4.6 Mha of shrubland (Table 1). In addition, urban expansion led to a loss of 24.3 Mha of cropland, producing the equivalent of 122 million tons (Mton) of cereals per year (Table 2). The amount of new cropland required to compensate for this loss in production

Institute for Environmental Studies, Vrije Universiteit Amsterdam, Amsterdam, the Netherlands. e-mail: jasper.van.vliet@vu.nl

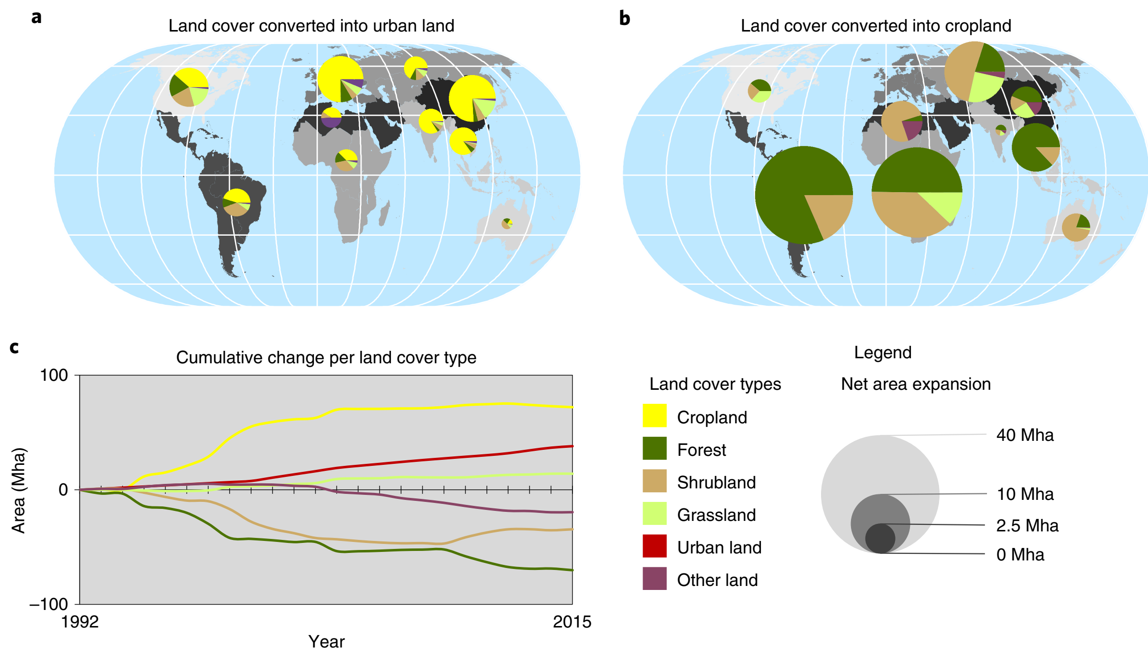


**Fig. 1 | Observed land cover changes between 1992 and 2015. a**,**b**, Land cover changes as a result of urban expansion and cropland expansion, respectively, for ten world regions included in this study. Regions are shown in the background in different shades of grey and in more detail in Supplementary Fig. 1. Pie charts are scaled according to net area increase in both classes between 1992 and 2015. Note that there was a net decrease in cropland in Europe, hence no pie chart is depicted. **c**, The cumulative net change for each of the six major land cover types at a global level.

**Table 1 | Direct land cover change as a result of urban land expansion between 1992 and 2015**

| Region | Urban expansion (Mha) | Direct land cover change due to urban expansion (Mha) | | | | |
|---|---|---|---|---|---|---|
| | | Forest | Shrubland | Cropland | Grassland | Other land |
| Canada and the United States | 6.1 | 1.2 | 1.4 | 2.3 | 1.2 | 0.1 |
| China | 8.8 | 0.2 | 0.5 | 6.7 | 1.2 | 0.1 |
| Europe | 8.4 | 0.7 | 0.9 | 6.3 | 0.5 | 0.5 |
| India | 2.4 | 0.1 | 0.2 | 2.0 | 0.1 | 0.0 |
| Latin America | 3.1 | 0.3 | 0.5 | 1.4 | 0.2 | 0.1 |
| MENA | 1.6 | 0.0 | 0.0 | 0.7 | 0.0 | 0.8 |
| Oceania | 0.5 | 0.1 | 0.1 | 0.1 | 0.1 | 0.0 |
| Russia and Central Asia | 2.2 | 0.2 | 0.3 | 1.5 | 0.2 | 0.1 |
| Southeast Asia | 3.0 | 0.2 | 0.2 | 2.5 | 0.0 | 0.1 |
| Sub-Saharan Africa | 1.9 | 0.3 | 0.5 | 0.7 | 0.2 | 0.1 |
| World total | 38.0 | 3.3 | 4.6 | 24.3 | 3.7 | 1.9 |

depended on where this new cropland was developed because cropland productivity differed between regions. Under the assumption that the cropland displacement took place within the same region, urban expansion led to 32.5 Mha of displaced cropland, globally (Supplementary Table 3). Under the assumption that the cropland displacement took place across all regions, urban expansion led to 58.0 Mha of displaced cropland, globally (Supplementary Table 4). Consistently, the loss of forest and shrubland as a result of cropland displacement also depended on where cropland was displaced to, because the percentage of new cropland resulting in a conversion of forest and shrubland differed between regions. Under the assumption that the cropland displacement took place within the same region, urban expansion led to 17.8 Mha of indirect forest loss and 7.0 Mha of indirect shrubland loss, whereas the remaining 7.6 Mha of displaced cropland led to a conversion of grassland and other land (Table 3 and Supplementary Table 3). Under the assumption that the cropland displacement took place across all regions, urban expansion led to 32.4 Mha of indirect forest loss and 17.4 Mha of indirect shrubland loss, whereas the remaining 8.2 Mha of displaced cropland led to a conversion of grassland and other land (Table 3 and Supplementary Table 4).

The ratio between the productivity of cropland converted into urban land and the productivity of new cropland required to compensate for this loss may be interpreted as a leverage factor. A leverage factor greater than 1 indicates that the cropland converted into urban land had a higher productivity than the displaced cropland. This comparison means that the area of new cropland required to compensate for the loss in crop production was greater than the amount of cropland area that was converted into urban land. Conversely, a leverage factor less than 1 indicates that the cropland converted into urban land had a lower productivity than the new and displaced cropland, which means that the area of

**Table 2 | Loss of crop production due to urban expansion and new cropland required to compensate for this loss between 1992 and 2015 under different assumptions of cropland displacement**

| Region | Loss of crop production due to urban expansion (Mton) | Cropland required to compensate (Mha) | | Leverage factor[a] (a.u.) | |
|---|---|---|---|---|---|
| | | Within-region displacement | Across-region displacement | Within-region displacement | Across-region displacement |
| Canada and the United States | 10.4 | 3.2 | 4.9 | 1.37 | 2.10 |
| China | 54.2 | 10.7 | 25.7 | 1.61 | 3.85 |
| Europe | 27.3 | 13.1 | 13.0 | 2.08 | 2.06 |
| India | 7.1 | 2.9 | 3.4 | 1.43 | 1.65 |
| Latin America | 3.3 | 1.6 | 1.5 | 1.13 | 1.11 |
| MENA | 1.0 | 0.7 | 0.5 | 1.10 | 0.74 |
| Oceania | 0.1 | 0.1 | 0.1 | 0.90 | 0.69 |
| Russia and Central Asia | 2.4 | 2.5 | 1.2 | 1.64 | 0.77 |
| Southeast Asia | 15.2 | 7.0 | 7.2 | 2.77 | 2.85 |
| Sub-Saharan Africa | 1.1 | 1.0 | 0.5 | 1.29 | 0.68 |
| World total | 122.1 | 32.5 | 58.0 | 1.34 | 2.39 |

[a]Leverage factors indicate the ratio between the productivity of cropland converted into urban land and the productivity of new cropland required to compensate for this loss.

new cropland required to compensate for the loss in crop production was lower than the amount of cropland area that was converted into urban land.

The assumption that cropland is displaced within the same world region led to a leverage factor of 1.34 for all world regions together (Table 2); that is, the area of new cropland that was required to compensate for the loss in crop production was 34% greater than the area of cropland that was lost to urbanization. Yet, leverage factors differed considerably between regions, ranging from 0.90 for Oceania to 2.77 for Southeast Asia. This means that new cropland in Oceania was more productive than cropland converted into urban land in Oceania, whereas new cropland in Southeast Asia was much less productive than cropland that converted into urban land in that region. The assumption that crop production lost to urban expansion was compensated across all world regions led to a global leverage factor of 2.39 (Table 2). This means that the area of new cropland that was required to compensate for the loss in crop production was 139% greater than the area of cropland that was lost to urbanization, for all world regions combined. On a regional level, a leverage factor of, for example, 2.06 for Europe means that the average productivity of cropland converted into urban land in Europe is 106% higher than the average productivity of all new croplands, globally.

The effect of displacement within regions compared with the displacement across regions differed between world regions. In India, for example, the leverage factor for displacement within the same region was 1.43 and the leverage factor for displacement across regions was 1.65 (Table 2). Thus, both assumptions led to a leverage factor greater than 1, indicating a leverage effect. In Russia and Central Asia, however, cropland displacement within the same region led to a leverage factor of 1.64, whereas cropland displacement across regions led to a leverage factor of 0.77. This difference in leverage factors indicated that cropland converted into urban land in Russia and Central Asia was on average 64% more productive than new cropland developed in the same region, but it was 23% less productive than new cropland developed in other regions. Globally, cropland displacement across all world regions led to a higher leverage factor and thus to a greater indirect loss of forest and shrubland than displacement within world regions. The difference was caused by a large amount of urban expansion in regions with relatively high cropland productivity, such as China, Canada and the United States, in combination with a large amount of cropland expansion in regions with relatively low average productivity, such as Sub-Saharan Africa and Latin America (Fig. 2). Because it was not possible to trace precisely where the cropland was displaced to, the results of both assumptions may be interpreted as the end points for a range of plausible results (Table 3).

**Impact of land use management.** To assess to what extent the leverage factor of cropland displacement was caused by differences in land use management and to what extent it was a result of different biophysical properties, the same analyses were repeated using potential yields instead of actual yields. Using the potential yields as a basis, the amount of displaced cropland decreased slightly to 28.0 Mha for displacement within world regions and 41.7 Mha for displacement across all regions (Supplementary Tables 5 and 6). Consequently, leverage factors for cropland displacement based on potential yield decreased to 1.15 to 1.72, as compared to 1.34 to 2.39 based on actual yields, on a global level. These numbers resulted in an indirect loss of 14.9 to 23.3 Mha of forest and 6.1 to 12.5 Mha of shrubland (Supplementary Tables 5 and 6 and Fig. 3). The lower and upper bounds of these values indicate that cropland displacement occurred within and across regions, respectively. Thus, when accounting for differences in land use management, the leverage effect in cropland displacement remained, and the indirect losses of forest and shrubland still exceeded the direct losses by a wide margin (Fig. 3).

The crops cultivated in the newly developed cropland areas were not necessarily the same as the crops previously cultivated in areas converted into urban land. To account for these differences in crop mixes, the analysis was repeated on the basis of the caloric values of the produce of a larger group of 16 different crop types. These crops included the 'cereal crops', wheat, maize and rice, but also crops that are often associated with tropical deforestation such as oil palm and soybean. Using actual yields, 29.8 to 43.7 Mha of new cropland were required to compensate for the lost production due to urban expansion, leading to a leverage factor of 1.23 to 1.80 (Supplementary Tables 7 and 8). This led to 15.0–24.4 Mha of indirect forest loss and 6.8–13.1 Mha of indirect shrubland loss (Supplementary Tables 7 and 8 and Fig. 3). The use of potential yields instead of actual yields to calculate caloric productivity decreased the leverage factor to a range between 1.11 and 1.49, globally, leading to 13.6–20.3 Mha of indirect forest loss and 6.0–10.9 Mha of indirect shrubland loss

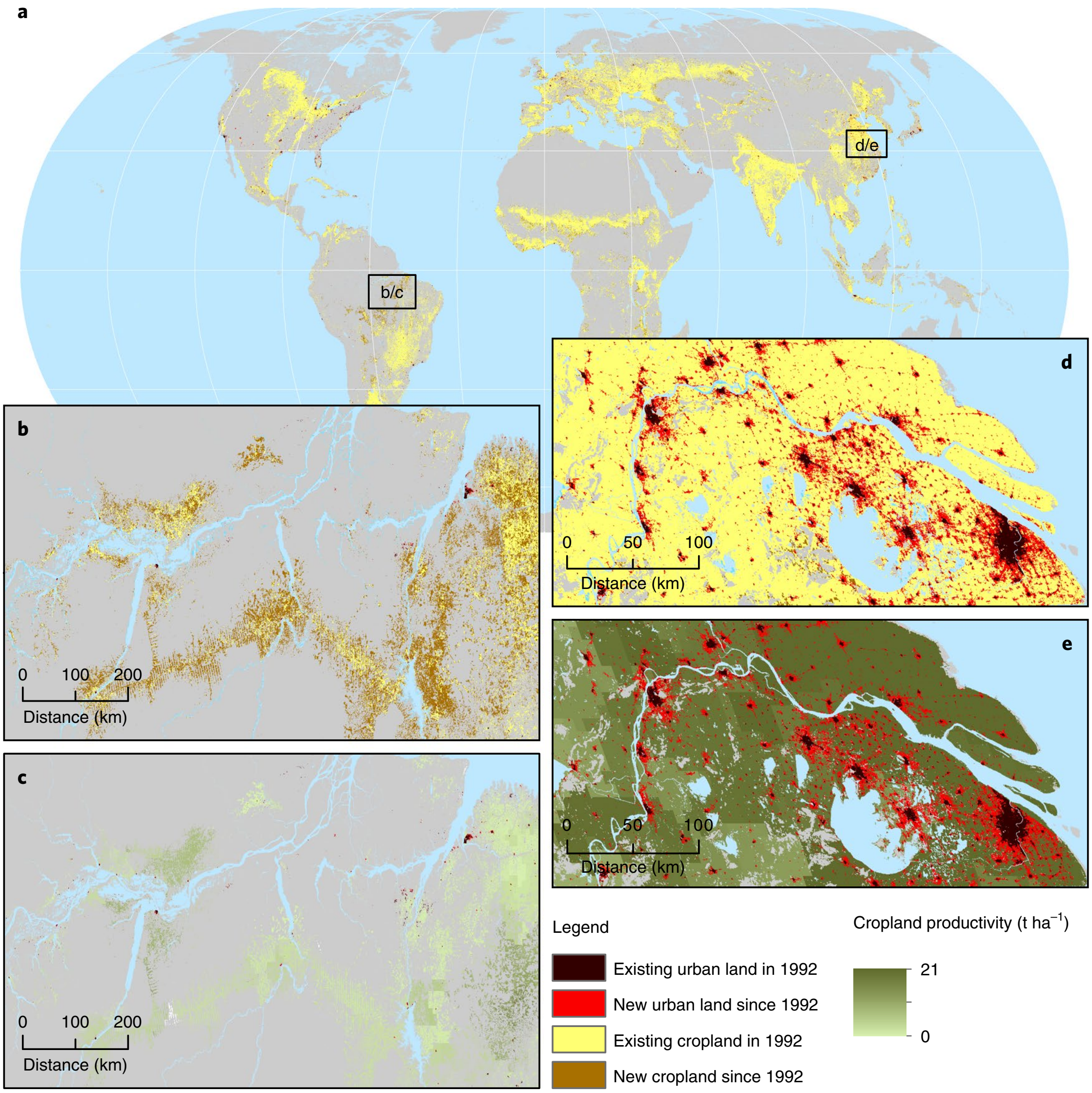


**Fig. 2 | Urban expansion and cropland expansion between 1992 and 2015. a**, The map shows locations of existing cropland in 1992, new cropland since 1992, existing urban land in 1992 and new urban land since 1992 at a global scale. **b–e**, The maps show close-ups of two typical but contrasting areas in terms of observed land use change and the cropland productivity in these areas. **b**, The map depicts part of the Amazon River basin characterized by a large amount of cropland expansion between 1992 and 2015, which is characterized by a low cropland productivity (**c**). **d**, The map shows Shanghai and the southern part of Jiangsu province in China, which has experienced a large amount of urban expansion between 1992 and 2015, mostly on highly productive cropland (**e**).

(Fig. 3, Supplementary Tables 9 and 10). Accounting for differences in cultivated crops thus resulted in lower leverage factors than the analysis that was based on major cereal crops only, but these leverage factors remained greater than 1. Moreover, indirect forest and shrubland losses also remained much greater than direct losses when accounting for different crop types (Fig. 3).

## Discussion

This study analysed the direct and indirect losses of forest and shrubland due to urban expansion, where indirect losses are a consequence of cropland displacement. The results show that urban expansion mostly leads to a direct loss of cropland, and new cropland mostly leads to a conversion of forest and shrubland elsewhere. As a result, indirect losses of forest and shrubland due to urban expansion are much greater than direct losses, confirming this hypothesis. Results also confirm the hypothesis that cropland displacement is leveraged by the differences in productivity of lost cropland and new cropland areas. Different assumptions for cropland displacement and cropland productivity affect the strength of the leverage effect and thus the size of indirect land cover changes, but both hypotheses remain confirmed under all the assumptions that are used.

Urban expansion also leads to a conversion of grassland. However, the global land cover data used in this study does not differentiate between managed grassland (that is, pastures) and natural grassland. When urban expansion leads to a conversion of natural grassland, these conversions may further add to the direct losses of natural area. However, when urban expansion leads to a conversion of pastures, that expansion may lead to pasture displacement. Such displacement may further add to the indirect losses of natural area from urban expansion given the important role of pasture expansion in deforestation, especially in the Amazon River basin[17,18]. As a consequence, both the direct and the indirect losses of natural area from urban expansion are probably higher than the

**Table 3 | Indirect loss of land cover as a result of cropland displacement due to urban expansion between 1992 and 2015 under different assumptions of cropland displacement**

| Region | Indirect forest loss (Mha) | | Indirect shrubland loss (Mha) | | Indirect grassland loss (Mha) | | Indirect loss of other land (Mha) | |
|---|---|---|---|---|---|---|---|---|
| | Within-region displacement | Across-region displacement | Within-region displacement | Across-region displacement | Within-region displacement | Across-region displacement | Within-region displacement | Across-region displacement |
| Canada and the United States | 1.3 | 2.7 | 0.8 | 1.5 | 1.2 | 0.5 | 0.0 | 0.2 |
| China | 5.3 | 14.1 | 1.9 | 7.6 | 2.7 | 2.7 | 1.7 | 0.9 |
| Europe | 7.9 | 7.1 | 3.4 | 3.8 | 1.3 | 1.4 | 0.4 | 0.5 |
| India | 1.3 | 1.9 | 0.7 | 1.0 | 0.5 | 0.4 | 0.3 | 0.1 |
| Latin America | 1.1 | 0.8 | 0.3 | 0.5 | 0.0 | 0.2 | 0.0 | 0.1 |
| MENA | 0.0 | 0.3 | 0.5 | 0.1 | 0.0 | 0.1 | 0.1 | 0.0 |
| Oceania | 0.0 | 0.0 | 0.1 | 0.0 | 0.0 | 0.0 | 0.0 | 0.0 |
| Russia and Central Asia | 0.5 | 0.6 | 1.2 | 0.3 | 0.6 | 0.1 | 0.1 | 0.0 |
| Southeast Asia | 5.0 | 4.0 | 0.7 | 2.1 | 0.0 | 0.8 | 0.0 | 0.3 |
| Sub-Saharan Africa | 0.5 | 0.3 | 0.4 | 0.1 | 0.1 | 0.1 | 0.0 | 0.0 |
| World total | 17.1 | 31.8 | 7.0 | 17.1 | 5.3 | 6.2 | 2.3 | 2.0 |

Values indicate the indirect land cover change related to urban expansion in the respective regions.

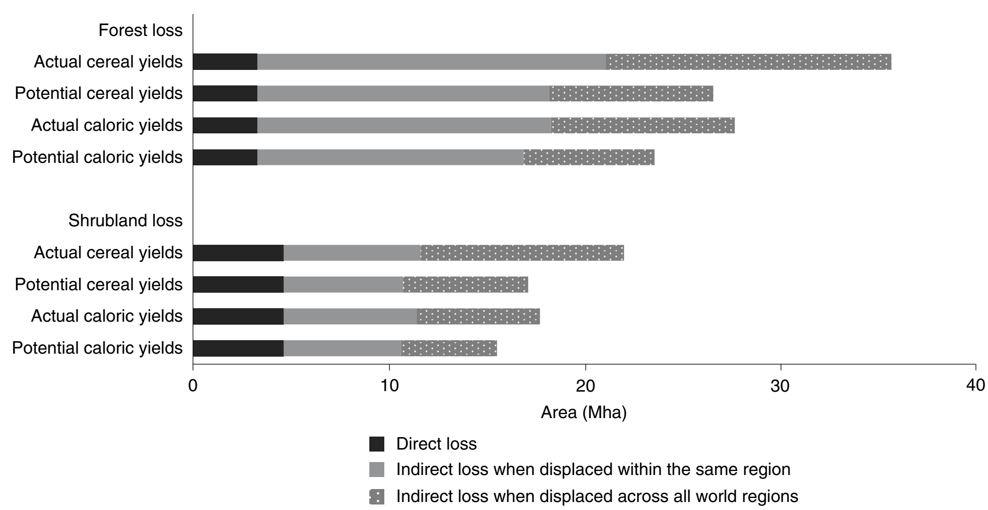


**Fig. 3 | Global loss of forest and shrubland as a result of urban expansion between 1992 and 2015 under different assumptions for cropland displacement.** The graph shows for both land cover types the direct changes into urban land and indirect changes as a result of cropland displacement. Direct changes are not affected by assumptions for cropland displacement, while indirect changes differ based on the crop types included in the calculation, the use of actual and potential yields, and whether cropland is assumed to be displaced within each world region or across all world regions. Details of the assumptions underlying cropland displacement are provided in the main text.

losses reported here, which are based only on the displacement of cropland.

Between 1992 and 2015, the total urban expansion was equal to 38.0 Mha, an area slightly larger than the land area of Japan. This change corresponds to an increase in urban land of 1.66 Mha per year. Two recently presented global datasets present yearly increases of urban land: (1) 1.19 Mha per year between 1975 and 2014 (ref. [19]) and (2) 1.49 Mha per year between 1990 and 2010 (ref. [20]). As both of these datasets present only the absence and presence of built-up land and involve no other land cover classes, it is not possible to assess to what extent those differences may affect the results presented in this study. Comparisons of global maps of urban land indicate that differences may arise due to different definitions, as well as different data sources, but that these differences are distributed equally over all world regions[21,22]. Therefore, differences between estimates may be to likely affect the size of the direct and indirect land cover change, but they may be unlikely to change the main findings of this paper: that indirect changes exceed direct changes and that this effect is leveraged by differences in cropland productivity.

Few studies have previously assessed the future impacts of indirect land changes resulting from urban expansion at local[14] and global[15] scales. The global assessment[15] projected a displacement of almost 65 Mton of crop production, corresponding to between 6.7 and 35 Mha of new cropland between 2000 and 2040. On the basis of the same quantification of crop production (actual yields of wheat, maize and rice), the present study finds 122 Mton of crop displacement corresponding to 32.5–58.0 Mha of new cropland in only 23 years. The difference between the previous study and the present study relates to the amount of urban expansion and the productivity of cropland areas. Because the present study is based on empirical data rather than model-based projections, this comparison suggests that the cropland displacement and indirect losses in forest and natural area until 2040 could far exceed the previously presented simulation results.

**Leverage from cropland displacement.** Global agricultural trade has increased in recent decades[23], especially between more affluent countries (where the majority of the urban expansion has taken place) and developing countries (where most of the new cropland is located[24]). China is a prime example of the process analysed in this study because it has experienced the greatest amount of urban expansion of all regions between 1992 and 2015, while the import of cropland products into China has increased rapidly in recent decades[25]. Because most urban expansion has taken place in developed regions with higher cropland productivity (see also Fig. 2c,d) and most cropland expansion has taken place in developing countries with lower cropland productivity (See also Fig. 2a,b), the leverage effect is likely to be on the higher side of the values presented for all regions combined.

The difference between the productivity of cropland converted to urban land and the productivity of new cropland is a result of both the biophysical suitability of locations and land management practices[26]. As land use intensity is typically higher around urban areas and lower in more remote areas[12], the differences in land use management could affect the leverage effect in cropland displacement. Based on actual yields, cropland displacement within the same region requires 34% more land than what was converted into urban land. Based on potential yields, cropland displacement within the same region requires 15% more land than what was converted into urban land. The 19% difference between the actual and potential yields can thus be attributed to land management. Similarly, based on actual yields, cropland displacement across all regions requires 139% more land than what was converted into urban land. On the basis of potential yields, cropland displacement across all regions requires 72% more land than what was converted into urban land. The 67% difference can thus be attributed to land use management; that is, land use management and biophysical suitability each have a roughly similar impact on the leverage effect of cropland displacement.

Land use intensification may also take place in existing croplands that are not affected by urban expansion, although spatially explicit data about this intensification is not available[27]. Yet, a recent review found that cropland expansion in forest frontiers is often market-driven, whereas management intensification is more often fuelled by technological development[28]. This finding suggests that losses in crop production from urban expansion mainly lead to the development of new cropland, as is assumed in the present study. In addition, cropland losses may increase the price for crop products because they increase land rents for the remaining cropland areas and thus also the prices for crop products produced in these remaining areas. This increase in price could lead to a decrease in the demand for crop products. However, price elasticities for agricultural commodities are low[29] and the loss in crop production due to urban expansion is small relative to the total crop production on a global scale. Therefore, this study assumes that (1) these effects do not alter the total demand for crop products and (2) all crop production lost to urban expansion is displaced elsewhere.

Crops associated with tropical deforestation are not necessarily the same crops that are cultivated on cropland converted into urban land. For example, palm oil is often associated with deforestation in tropical areas[30,31], but they are rarely found in areas with urban expansion. Another example is the rapid expansion of soybean cultivation, such as in the Gran Chaco in Argentina and in other areas in the Neotropics[32,33]. However, accounting for differences in crop mixes hardly changes the leverage effect or the indirect losses in forest and shrubland as a result of cropland displacement. Hence, this finding reinforces the confirmation of both hypotheses tested in this study and indicates that the results are not due to specific crop types.

**Implications.** Urban areas are expected to continue expanding in the next decades[34,35], and recent projections indicate that this expansion may severely impact food production[36] as well as natural area[37]. This expected urban expansion offers a window of opportunity to guide urban development trajectories to minimize environmental impact. Solutions to reduce the competition for land often focus on agriculture[7,8,26], whereas the potential for a more efficient use of urban areas have hardly been explored.

The results of this study suggest that both the location and the total area of urban expansion provide opportunities to reduce the competition for land. Instead of converting fertile cropland, urban development could be directed towards less productive areas. Because urban areas are increasingly decoupled from their resource base[38], such allocation decisions may now be made without greatly compromising their functionality. The economic and intensive land use policy in China may be a step in this direction as it aims to protect specifically the most fertile cropland against urban expansion[39]. Additionally, a focus on urban densification may reduce the amount of urban expansion. Currently, major differences exist in the population densities of cities across different continents[34,40], indicating that there is room to reduce the built-up land area per person in many world regions. A few countries, including China and some in Europe, have already introduced policies that promote compact cities or reduce urban sprawl to steer urban development trajectories towards reducing their environmental impact[41,42]. At the same time, examples from the United States and Australia show that not all planning initiatives towards compact development have been effective[43,44]. Urban planning outcomes in regions of the Global South (low- and middle-income regions of Asia, Africa, Latin America and the Caribbean) are further challenged by conflicting realities; that is, they are caught between the policies of governments and administrations on one side and the realities of survival of the poor and often marginalized populations on the other[45]. Moreover, compact urban development has to be balanced against other dimensions of sustainable urbanization to preserve the livability of cities for their inhabitants[46].

Land use and land use change are pivotal to many sustainability challenges, and model-based assessments are elementary in analysing possible solutions[47,48]. Yet, the representation of urban systems in land use models is rather simplistic in contrast to the modelling of agricultural and natural land systems[49,50]. As a consequence, the potential of alternative urban development trajectories to mitigate direct and indirect land cover change, as well as related environmental impacts, remains to be investigated.

## Methods

This study calculated the direct and indirect changes in forest and shrubland as a result of urban expansion. Direct changes refer to the conversion of forest and shrubland into urban land. Indirect changes refer to the conversion of forest and shrubland into cropland to compensate for the conversion of cropland into urban land elsewhere; that is, indirect changes were the result of cropland displacement. Changes in land cover and crop production were analysed at the pixel level and subsequently aggregated to the scale of ten separate world regions (Canada and United States, China, Europe, India, Latin America, MENA, Oceania, Russia and Central Asia, Southeast Asia and Sub-Saharan Africa) as well as the entire world. The aggregation to regions was required because of the need to relate pixels where cropland converted into urban land to other pixels where new cropland appeared and thus calculate the cropland that was displaced by urban expansion.

The regions in this analysis were delineated on the basis of the standard regions used by the World Bank, with the notable exception of (1) East Asia and Pacific and (2) Europe and Central Asia. These regions were further subdivided to reflect the differences in both urban expansion and agricultural trade dynamics. Specifically, the East Asia and Pacific region was further subdivided into China, Southeast Asia and Oceania. This subdivision follows analyses of agricultural trade flows[24,51], which indicated that China was increasingly importing cropland products, whereas Southeast Asia is a major source of those cropland products[24,25,52]. Similarly, Europe was separated from Russia and Central Asia because Europe had experienced a greater amount of urban expansion, but this development was much less pronounced in Russia and Central Asia. As a result, the regions represent relatively coherent groups of countries for which the majority of the crop products consumed are also produced within that region[51], which justifies the assumption of a within-region displacement. At the same time, these regions also represent regions that form the sending and receiving side of the rapidly increasing amount of global trade in crop products. The regions are depicted in Supplementary Fig. 1.

**Land cover change analysis.** The land cover changes between 1992 and 2015 were based on the ESA CCI Land Cover data, which provides land cover maps for all years between 1992 to 2015 (ref. [16]). The land cover data in this dataset were derived from multiple sensors and presented at an approximately 300-m resolution (depending on the latitude)[53]. All land cover maps were reclassified into six aggregate classes (cropland, forest, urban land, shrubland, grassland and other). The land cover mosaic classes in the ESA CCI data were reclassified into combinations of the aggregated classes according to the shares of the respective plant functional types found in these mosaic classes[54]. Urban land in this study corresponds to the class 'urban areas' on the ESA CCI map. The classification of the ESA CCI follows the United Nations' Land Cover Classification System (LCCS)[55], which defines urban areas as "primarily non-vegetated areas with an artificial cover resulting from human activities"[56]. The complete reclassification scheme is presented in Supplementary Table 11. Reclassified land cover maps were combined with an areal grid to account for different cell sizes to find the total area for each land cover and for each year.

Land cover maps for the years 1992 and 2015 were overlain to derive a land cover change map, indicating—for each pixel—the land cover at the start and at the end of the study period. This land cover change map was also combined with an areal grid to obtain the total area per land cover change type between 1992 and 2015. All pixels that were classified as either forest or shrubland in 1992 and changed into urban land in 2015 were considered direct land cover changes as a result of urban expansion, and their areas were calculated accordingly.

**Quantification of cropland displacement.** To calculate cropland displacement, the total crop production of pixels converted into urban land was calculated and used to compute the equivalent amount of newly developed cropland required to produce the same amount of crops. Both new cropland and cropland converted into urban land were derived from the land cover change between 1992 and 2015. Crop production, rather than cropland area, was used for this analysis to account for the differences in productivity in different locations. To compare pixels with different crop types, a representative productivity was calculated for each pixel, which was the productivity that would be obtained when all harvested area was covered with wheat, maize or rice, proportional to the actual occurrence of wheat, maize and rice in that location. Together these three cereal crops represented about 65% of all harvested area globally, and at least one of these three crops may be found in the vast majority of all cropland areas around the world. Therefore, and because their yields under favourable conditions were comparable, this operationalization of productivity was deemed suitable to calculate cropland displacement[57,58].

The representative productivity (tons of produce per hectare of cropland) was calculated at a 5-arcminute resolution by multiplying the average yield of wheat, maize and rice (tons per hectare of harvested area) with the multi-cropping factor in each pixel. The average yield was calculated as the area-weighted average of the yields of these three crops, where areas refer to the harvested area of each of these three crops in that pixel. The multi-cropping factor was calculated as the total harvested area of all 175 crops covered in Monfreda et al.[59] in a pixel divided by the cropland area in that same pixel as reported in Ramankutty et al.[60]. Yields for wheat, maize and rice were taken from Mueller et al.[61], which provided data for around the year 2000. This yield data is an updated yet consistent version of the yields presented in Monfreda et al.[59] and thus is also consistent with the calculation of the multi-cropping factor. Gaps in the spatial coverage (that is, pixels for which no wheat, maize, or rice data were estimated in the Monfreda data, but for which either a cropland loss or cropland gain was reported in the ESA CCI land cover data) were filled using a focal average (that is, the average of all pixels directly and diagonally adjacent to this location). Any remaining gaps were not filled; instead, these locations were excluded from the calculation of average productivity of cropland changes. These gaps represent 2.4% of all cropland converted into urban land and 1.9% of all new cropland areas. As these percentages were close, this decision was unlikely to yield any systematic bias towards either of these change types. After this data processing, productivity data was resampled to the resolution of the land cover maps using a nearest neighbour assignment. Supplementary Table 12 provides more detail of all data used in this study.

Cropland displacement was calculated using two contrasting assumptions: either cropland was displaced within the same region or cropland was displaced across all regions. In both cases, cropland displacement was calculated as the amount of cropland that was required to compensate for the loss in crop production due to urban expansion. For cropland displacement within a region, displacement was based on the average productivity of all new cropland that appeared within that same region between 1992 and 2015. For cropland displacement across all regions, displacement was based on the average productivity of all new cropland that appeared in all regions between 1992 and 2015. These two situations were reported as extreme values that bound the possibility space of cropland displacement.

For Europe, it was not possible to completely compensate for all lost crop production within the same region because the amount of newly developed cropland between 1992 and 2015 was not large enough. Therefore, under the assumption of displacement within the region, the amount of crop production that could be compensated for by new cropland in Europe was displaced within the region, whereas the additional crop production that could not be compensated for within Europe was displaced across all other world regions.

**Actual and potential yields.** The analyses of cropland displacement were conducted for actual yields as well as for potential yields on the basis of data for around the year 2000 (ref. [61]) (Supplementary Table 12). Potential yield is defined here as the attainable yield after water and nutrient deficiencies have been removed and serves as a way to separate the impact of land management from the inherent biophysical suitability of locations to produce crops. The results based on potential yield thus indicated cropland displacement if the newly developed land was managed with the same intensity as the cropland converted into urban land.

Newly developed cropland areas may differ from cropland converted into urban land in the mix of crops that is grown. These differences were not necessarily reflected in the average productivity of wheat, maize and rice. Therefore, the calculations were repeated using the actual and potential productivity of a larger number of crops, expressed in caloric value. This calculation was based on 16 food crops for which both actual and potential yield information was available from Mueller et al.[61]; that is wheat, rice, maize, soybeans, barley, sorghum, millet, rapeseed, groundnut, sunflower, sugarcane, potato, cassava, palm oil, rye and sugar beet. These crops coincide with the crop types used in an earlier analysis of cropland losses from urban expansion[36]. Together, these crops represented 76% of all harvested area globally, including some of the crops that have been associated with deforestation in recent years, such as palm oil and soybean[30,32]. The analysis based on these 16 crops was otherwise similar to the analysis based on the productivity of the three major cereal crops in that a representative productivity was calculated on the basis of the area-weighted average productivity of the crops included, except that productivity was expressed in thousands of calories rather than tons of produce. Crop yields were converted into caloric values using standard nutritive values as reported for the different crops by the FAO[62].

**Indirect loss of forests and shrubland.** Displaced cropland as a result of urban expansion was multiplied by the percentage of new cropland leading to a loss of forests and shrubland, respectively, to calculate indirect losses for forests and shrubland. For cropland displacement within the region, these numbers indicated the percentage of new cropland leading to a conversion of forests and shrublands within that region. For cropland displacement across all regions, these numbers indicated the percentage of new cropland leading to a conversion of forests and shrublands in all regions. Consistent with the reporting of cropland displacement, indirect losses of forest and shrubland were reported as a range of values bound by the assumptions of displacement within the same region and displacement across all regions.

**Leverage factors.** To further express the impact of cropland displacement, a leverage effect was calculated as $P_{converted}/P_{new}$, where $P_{converted}$ is the average productivity of cropland converted into urban land, and $P_{new}$ is the average productivity of new cropland areas that may be used to compensate for the loss in crop production. Values greater than 1 indicated that the area of cropland required to compensate for the lost production due to urban expansion exceeded the area of cropland that was lost, and values less than 1 indicate the opposite. Indirect losses of forest were subsequently calculated on the basis of the amount of displaced cropland and the share of new cropland leading to conversion of forest. Similarly, indirect losses of shrubland were calculated on the basis of the amount of displaced cropland and the share of new cropland leading to conversion of shrubland.

**Implementation.** The area of each cell in the ESA CCI land cover maps was derived from the area() function of the 'raster' package in R software. All other spatial analyses were implemented in Python, using spatial analysis functions from the ArcPy package. These spatial analyses provided results per world region. All spatial analysis results were post-processed in Microsoft Excel. This post-processing included combining regional results to obtain global-scale results.

## Data availability

Data that support the findings presented in this study are available from the author upon reasonable request.

## Code availability
Scripts used for this analysis are available from the author upon reasonable request.

Received: 15 November 2018; Accepted: 18 June 2019;
Published online: 29 July 2019

## References

1. Lambin, E. F. & Meyfroidt, P. Global land use change, economic globalization, and the looming land scarcity. *Proc. Natl Acad. Sci. USA* **108**, 3465–3472 (2011).
2. Newbold, T. et al. Global effects of land use on local terrestrial biodiversity. *Nature* **520**, 45–50 (2015).
3. Costanza, R. et al. Changes in the global value of ecosystem services. *Global Environ. Change* **26**, 152–158 (2014).
4. Tubiello, F. N. et al. The contribution of agriculture, forestry and other land use activities to global warming, 1990–2012. *Glob. Change Biol.* **21**, 2655–2660 (2015).
5. Curtis, P. G., Slay, C. M., Harris, N. L., Tyukavina, A. & Hansen, M. C. Classifying drivers of global forest loss. *Science* **361**, 1108–1111 (2018).
6. Geist, H. J. & Lambin, E. F. Proximate causes and underlying driving forces of tropical deforestation. *Bioscience* **52**, 143–150 (2002).
7. Alexander, P. et al. Drivers for global agricultural land use change: the nexus of diet, population, yield and bioenergy. *Global Environ. Change* **35**, 138–147 (2015).
8. Erb, K.-H. et al. Exploring the biophysical option space for feeding the world without deforestation. *Nat. Commun.* **7**, 11382 (2016).
9. Rudel, T. K. et al. Agricultural intensification and changes in cultivated areas, 1970–2005. *Proc. Natl Acad. Sci. USA* **106**, 20675–20680 (2009).
10. DeFries, R. S., Rudel, T., Uriarte, M. & Hansen, M. Deforestation driven by urban population growth and agricultural trade in the twenty-first century. *Nat. Geosci.* **3**, 178–181 (2010).
11. Meyfroidt, P., Lambin, E. F., Erb, K.-H. & Hertel, T. W. Globalization of land use: distant drivers of land change and geographic displacement of land use. *Curr. Opin. Env. Sust.* **5**, 438–444 (2013).
12. Avellan, T., Meier, J. & Mauser, W. Are urban areas endangering the availability of rainfed crop suitable land? *Remote Sens. Lett.* **3**, 631–638 (2012).
13. Gibbs, H. K. et al. Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. *Proc. Natl Acad. Sci. USA* **107**, 16732–16737 (2010).
14. Ke, X. et al. Direct and indirect loss of natural habitat due to built-up area expansion: a model-based analysis for the city of Wuhan, China. *Land Use Policy* **74**, 231–239 (2018).
15. van Vliet, J., Eitelberg, D. A. & Verburg, P. H. A global analysis of land take in cropland areas and production displacement from urbanization. *Global Environ. Change* **43**, 107–115 (2017).
16. European Space Agency Climate Change Initiative *Land Cover* http://maps.elie.ucl.ac.be/CCI/viewer/index.php (2015).
17. Armenteras, D., Espelta, J. M., Rodríguez, N. & Retana, J. Deforestation dynamics and drivers in different forest types in Latin America: three decades of studies (1980–2010). *Global Environ. Chang.* **46**, 139–147 (2017).
18. Barona, E., Ramankutty, N., Hyman, G. & Coomes, O. T. The role of pasture and soybean in deforestation of the Brazilian Amazon. *Environ. Res. Lett.* **5**, 024002 (2010).
19. Pesaresi, M. et al. *GHS Built-up Grid, Derived from Landsat, Multitemporal (1975, 1990, 2000, 2014)* (European Commission, Joint Research Centre, accessed 1 January 2017); http://data.europa.eu/89h/jrc-ghsl-ghs_built_ldsmt_globe_r2015b
20. Liu, X. et al. High-resolution multi-temporal mapping of global urban land using Landsat images based on the Google Earth engine platform. *Remote Sens. Environ.* **209**, 227–239 (2018).
21. Liu, Z., He, C., Zhou, Y. & Wu, J. How much of the world's land has been urbanized, really? A hierarchical framework for avoiding confusion. *Landscape Ecol.* **29**, 763–771 (2014).
22. Potere, D. & Schneider, A. A critical look at representations of urban areas in global maps. *GeoJournal* **69**, 55–80 (2007).
23. D'Odorico, P., Carr, J. A., Laio, F., Ridolfi, L. & Vandoni, S. Feeding humanity through global food trade. *Earths Future* **2**, 458–469 (2014).
24. Weinzettel, J., Hertwich, E. G., Peters, G. P., Steen-Olsen, K. & Galli, A. Affluence drives the global displacement of land use. *Global Environ. Change* **23**, 433–438 (2013).
25. Qiang, W., Liu, A., Cheng, S., Kastner, T. & Xie, G. Agricultural trade and virtual land use: the case of China's crop trade. *Land Use Policy* **33**, 141–150 (2013).
26. Foley, J. A. et al. Solutions for a cultivated planet. *Nature* **478**, 337–342 (2011).
27. Kuemmerle, T. et al. Challenges and opportunities in mapping land use intensity globally. *Curr. Opin. Env. Sust.* **5**, 484–493 (2013).
28. Byerlee, D., Stevenson, J. & Villoria, N. Does intensification slow crop land expansion or encourage deforestation? *Glob. Food Secur.-Agr.* **3**, 92–98 (2014).
29. Tabeau, A., Helming, J. & Philippidis, G. *Land Supply Elasticities* (Publications Office of the European Union, 2017).
30. Vijay, V., Pimm, S. L., Jenkins, C. N. & Smith, S. J. The impacts of oil palm on recent deforestation and biodiversity loss. *PLoS ONE* **11**, e0159668 (2016).
31. Meyfroidt, P. et al. Multiple pathways of commodity crop expansion in tropical forest landscapes. *Environ. Res. Lett.* **9**, 074012 (2014).
32. Gasparri, N. I., Grau, H. R. & Gutiérrez Angonese, J. Linkages between soybean and neotropical deforestation: coupling and transient decoupling dynamics in a multi-decadal analysis. *Global Environ. Change* **23**, 1605–1614 (2013).
33. Fehlenberg, V. et al. The role of soybean production as an underlying driver of deforestation in the South American Chaco. *Global Environ. Change* **45**, 24–34 (2017).
34. Angel, S., Parent, J., Civco, D. L., Blei, A. & Potere, D. The dimensions of global urban expansion: estimates and projections for all countries, 2000–2050. *Prog. Plann.* **75**, 53–107 (2011).
35. Fragkias, M., Güneralp, B., Seto, K. C. & Goodness, J. A. in *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities* (eds Elmqvist T. et al.) 409–435 (Springer Netherlands, 2013).
36. Bren d'Amour, C. et al. Future urban land expansion and implications for global croplands. *Proc. Natl Acad. Sci. USA* **114**, 8939–8944 (2017).
37. Seto, K. C., Guneralp, B. & Hutyra, L. R. Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proc. Natl Acad. Sci. USA* **109**, 16083–16088 (2012).
38. Cumming, G. S. et al. Implications of agricultural transitions and urbanization for ecosystem services. *Nature* **515**, 50–57 (2014).
39. Liu, Y., Fang, F. & Li, Y. Key issues of land use in China and implications for policy making. *Land Use Policy* **40**, 6–12 (2014).
40. Schneider, A. & Woodcock, C. E. Compact, dispersed, fragmented, extensive? A comparison of urban growth in twenty-five global cities using remotely sensed data, pattern metrics and census information. *Urban Stud.* **45**, 659–692 (2008).
41. Chen, H., Jia, B. & Lau, S. S. Y. Sustainable urban form for Chinese compact cities: challenges of a rapid urbanized economy. *Habitat Int.* **32**, 28–40 (2008).
42. Cortinovis, C., Haase, D., Zanon, B. & Geneletti, D. Is urban spatial development on the right track? Comparing strategies and trends in the European Union. *Landscape Urban Plan.* **181**, 22–37 (2019).
43. Boyle, R. & Mohamed, R. State growth management, smart growth and urban containment: a review of the US and a study of the heartland. *J. Environ. Plann. Man.* **50**, 677–697 (2007).
44. Bunker, R. How Is the compact city faring in Australia? *Plan. Pract. Res.* **29**, 449–460 (2014).
45. Watson, V. Seeing from the south: refocusing urban planning on the globe's central urban issues. *Urban Stud.* **46**, 2259–2275 (2009).
46. Westerink, J. et al. Dealing with sustainability trade-offs of the compact city in peri-urban planning across European city regions. *Eur. Plan. Stud.* **21**, 473–497 (2013).
47. Pouzols, F. M. et al. Global protected area expansion is compromised by projected land-use and parochialism. *Nature* **516**, 383–386 (2014).
48. Michetti, M. & Zampieri, M. Climate–human–land interactions: a review of major modelling approaches. *Land* **3**, 793–833 (2014).
49. Levis, S. Modeling vegetation and land use in models of the Earth System. *WIREs Clim. Change* **1**, 840–856 (2010).
50. van Vliet, J., Verburg, P. H., Grădinaru, S. R. & Hersperger, A. M. Beyond the urban-rural dichotomy: towards a more nuanced analysis of changes in built-up land. *Comput. Environ. Urban.* **74**, 41–49 (2019).
51. Kastner, T., Erb, K.-H. & Haberl, H. Rapid growth in agricultural trade: effects on global area efficiency and the role of management. *Environ. Res. Lett.* **9**, 034015 (2014).
52. Yu, Y., Feng, K. & Hubacek, K. Tele-connecting local consumption to global land use. *Global Environ. Change* **23**, 1178–1186 (2013).
53. Poulter, B. et al. Plant functional type classification for earth system models: results from the European Space Agency's land cover climate change initiative. *Geosci. Model Dev.* **8**, 2315–2328 (2015).
54. Li, W. et al. Gross and net land cover changes in the main plant functional types derived from the annual ESA CCI land cover maps (1992–2015). *Earth Syst. Sci. Data* **10**, 219–234 (2018).
55. European Space Agency Climate Change Initiative *Land Cover CCI Product User Guide v.2.0* http://maps.elie.ucl.ac.be/CCI/viewer/download/ESACCI-LC-Ph2-PUGv2_2.0.pdf (2017).
56. Di Gregorio, A. *Land Cover Classification System (LCCS); Classification Concepts and User Manual; Software v.2. Environment and Natural Resources Series No. 8* (Food and Agriculture Organization of the United Nations, 2005).
57. Neumann, K., Verburg, P. H., Stehfest, E. & Müller, C. The yield gap of global grain production: a spatial analysis. *Agr. Syst.* **103**, 316–326 (2010).
58. van Asselen, S. & Verburg, P. H. A land system representation for global assessments and land-use modeling. *Glob. Change Biol.* **18**, 3125–3148 (2012).

59. Monfreda, C., Ramankutty, N. & Foley, J. A. Farming the planet: 2. geographic distribution of crop areas, yields, physiological types, and net primary production in the year 2000. *Glob. Biogeochem. Cycles* **22**, GB1022 (2008).
60. Ramankutty, N., Evan, A. T., Monfreda, C. & Foley, J. A. Farming the planet: 1. geographic distribution of global agricultural lands in the year 2000. *Glob. Biogeochem. Cycles* **22**, GB1003 (2008).
61. Mueller, N. D. et al. Closing yield gaps through nutrient and water management. *Nature* **490**, 254–257 (2012).
62. *Nutritive Factors* (Food and Agriculture Organization of the United Nations, accessed 27 February 2019); http://www.fao.org/fileadmin/templates/ess/ess_test_folder/Food_security/Excel_sheets/Nutritive_Factors.xls

## Acknowledgements
J.v.V. thanks R. Prestele for his help with the spatial data analysis. This project was supported by NWO-WOTRO project no. W 07.303.108 on joint SDG research. This paper contributes to the Global Land Programme (https://glp.earth).

## Author contributions
J.v.V. designed the study, conducted the data analysis and wrote the paper.

## Competing interests
The author declares no competing interests.

## Additional information
**Supplementary information** is available for this paper at https://doi.org/10.1038/s41893-019-0340-0.

**Reprints and permissions information** is available at www.nature.com/reprints.

**Correspondence and requests for materials** should be addressed to J.v.V.

**Publisher's note:** Springer Nature remains neutral with regard to jurisdictional claims in published maps and institutional affiliations.