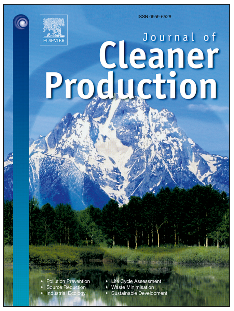


The processes of land use change in mining regions

L.J. Sonter, C.J. Moran, D.J. Barrett, B.S. Soares-Filho

PII: S0959-6526(14)00317-5
DOI: 10.1016/j.jclepro.2014.03.084
Reference: JCLP 4191

To appear in: *Journal of Cleaner Production*

Received Date: 30 October 2013
Revised Date: 24 March 2014
Accepted Date: 24 March 2014

Please cite this article as: Sonter LJ, Moran CJ, Barrett DJ, Soares-Filho BS, The processes of land use change in mining regions, *Journal of Cleaner Production* (2014), doi: 10.1016/j.jclepro.2014.03.084.

Title: The processes of land use change in mining regions

Authors: L. J. Sonter [a*], C. J. Moran [a], D. J. Barrett [ab], B. S. Soares-Filho[c]

Author affiliations:

[a]Sustainable Minerals Institute, The University of Queensland, Brisbane, Australia

[b]CSIRO Land and Water, Canberra, Australia

[c]Centro de Sensoriamento Remoto, Universidade Federal de Minas Gerais, Belo Horizonte, Brazil

*Corresponding author: l.sonter@uq.edu.au; phone: +61 7 3346 4027

Keywords: Atlantic Forest; Iron Quadrangle; mining; land change science; *Quadrilátero Ferrífero*; remote sensing; resource regions; sustainable development; teleconnections; time series.

Word count: 7149

Processes of land use change in mining regions

Abstract: The world's mining regions undergo abrupt and extensive land use change, the impacts of which pose significant management challenges for mining companies and regulatory agencies. In this study we investigated 20 years of land use change in Brazil's largest iron ore mining region, the *Quadrilátero Ferrífero* (QF) using a remote sensing classification procedure to produce a time series of land use maps and a Land Change analysis to investigate the causes and consequences of observed changes. The QF has undergone extensive land use change including deforestation, plantation expansion, urbanization and mine expansion. Comparing our results with those found in surrounding non-mining landscapes illustrated some important differences. For example, the QF contained additional highly profitable land uses, including mining and plantation forestry, which were driven by globalized markets for mineral resources. This finding suggests the processes of land use change within mining regions are distinct from those found elsewhere and, as such, land management policies and approaches should reflect this. We also identified four potential generalizations regarding these processes: 1) the direct footprint of mining expands over time, 2) the offsite footprint of mining is extensive and also often expanding, 3) the direct and indirect use of land by mining causes environmental and social impacts, some of which are not captured by current management approaches, and 4) the footprints of mining and their associated impacts are driven by global factors, many of which are uncontrollable by local land holders and regional management plans and policies. We describe and discuss these generalizations, drawing on published evidence from other mining regions to illustrate their generality and their implications for land management.

Keywords: Atlantic Forest; Iron Quadrangle; mining; land change science; *Quadrilátero Ferrífero*; remote sensing; resource regions; sustainable development; teleconnections; time series.

1. INTRODUCTION

Mining regions (also known as resource regions) are geologically defined by an abundance of economically feasible mineral resources and, as a result, they often undergo abrupt and extensive changes in land use (Bridge 2004). Land use change can be caused by a combination of mining and non-mining activities (Moran et al. 2013), both of which have environmental and social impacts. While these impacts are often negative, including land degradation, biodiversity loss (Simmons et al. 2008; Townsend et al. 2009) and livelihood displacement (Schueler et al. 2011); positive impacts can also occur, such as increased conservation activities and water quality management (Sonter et al. 2014; Sonter et al. 2013a). Managing the impacts of land use change—i.e., mitigating negative impacts and enhancing positive impacts—is an important sustainable development goal that poses a challenge for mining companies and regulatory agencies alike.

Despite this, little work has been done to understand the processes of land use change in mining regions. While some case study evidence has been presented describing how change has occurred in specific sites (e.g. Hammond et al. 2007), these studies often lack a rigorous framework to allow application of knowledge to other mining regions for the purpose of decision making. One recently proposed framework is that of Franks et al. (2013), which has been developed to analyze the cumulative impacts of mining at the regional scale. This framework, however, is not spatially and temporally explicit, which is essential in understanding processes of land use change and without this it is difficult to predict future land use change, identify and quantify potential tradeoffs in land management decisions, and develop policies capable of avoiding undesirable trajectories (Reid et al. 2006).

The field of Land Change Science presents an opportunity to overcome this limitation by analyzing spatially and temporally explicit processes of land use change. Using a Land Change approach, *land use* represents the interaction between humans and their environment and is used as a conceptual platform upon which to determine both the causes and consequence of land use change and to investigate the influence and potential success of land management decisions (Turner et al. 2007). To our knowledge only a few studies have used a Land Change analysis to investigate processes of land use change in mining regions (Schueler et al. 2011; Sonter et al. in review). None, however, have made comparisons with non-mining regions to examine their conceptual differences, nor have they made comparisons with other mining regions to investigate potential generalizations. The ability to make comparisons, generalize and extrapolate convincingly is necessary if frameworks, like that proposed by Franks et al. (2013), are to be helpful beyond their intellectual (or conceptual) value. It is also necessary to enable land management approaches be developed based on the evidence and experience learnt from other mining regions.

Analyzing land use change requires a time series of land use maps and remote sensing classification is the primary tool used to acquire such data (Lambin & Linderman 2006). There are many advantages of using remote sensing classification tools to map land use, e.g. it allows efficient access to otherwise inaccessible or remote locations and it provides time series information at a scale meaningful for regional decision making. While remote sensing has been used for a long time to monitor specific mining activities (e.g. Irons et al. 1980), few regional-scale analyses explicitly incorporate mining as a separate land use. Generally this is because mining operations occur at a small spatial scale relative to other land use changes (such as agricultural expansion and deforestation) and because performing regional classification at this scale can be a time-consuming and potentially inaccurate task (Sonter et al. 2013b). For this reason, mining is commonly merged into other land use classes, such as ‘cleared land’, ‘built-up land’ or ‘other’.

In this study we investigated 20 years of land use change within a large and well-established mining region: Brazil’s *Quadrilátero Ferrífero* (QF; Iron Quadrangle). We had two specific objectives. First, to quantify land use change within the QF to determine if these processes can be efficiently and accurately characterized using remotely sensed data, classification tools and a Land Change analysis. Second, to compare the processes observed within the QF with published information from surrounding non-mining landscapes to determine if the presence of mineral resources and a well-established mining industry creates fundamental differences in the processes of land use change than may otherwise be expected. Interpreting these results allowed us to hypothesize conceptual generalizations that may occur in other mining regions and we discuss the implications of these, drawing on published literature.

2. METHODS

2.1 Study region

The QF mining region covers approximately 1.98 Mha of land within the Atlantic Forest biome and the State of Minas Gerais (CODEMIG 2010; Figure 1). It has a long and important mining history, containing approximately 75% of Brazil’s measured iron ore reserve, half of which is graded above 60% iron content (Gurmendi 2011). The region also contains economically feasible gold and bauxite deposits, which are both also mined. Over the past two decades, the mining industry within the region has responded rapidly to the growing global demand for iron and steel (Sonter et al. in review). During this time productive capacity has tripled (MME 2011) making the QF the largest iron ore production and exportation region in Latin America. In regards to land use, most land in the region is currently under some form of mining tenure, including exploration, pre-operational or approved land for mining (DNPM 2012).

Land in the QF is also used for other non-mining purposes, including biodiversity conservation, water resources management, plantation forestry and urban development (Jacobi & do Carmo 2008; Sonter et al. 2014). The impacts of mining have been shown to heavily influence these adjacent land users, through pollution (Matschullat et al. 2000) and by influencing other land use opportunities (Sonter et al. in review). In response to growing concerns surrounding industry operation and the potential for land use conflicts in the near future, the State of Minas Gerais has undergone significant regulatory change regarding environmental licensing and rehabilitation requirements (Viana & Bursztyn 2010). In addition, the surrounding Atlantic Forest biome has been subject to regulatory changes regarding forest management, given the biome's dwindling forest remnants (Ribeiro et al. 2009).

2.2 Remote sensing classification of land use change

Landsat TM data was chosen for analysis because its spatial and temporal scale (resolution and extent) allowed mining operations to be identified (Irons et al. 1986). Performing classification at a 30 m spatial resolution was also important for accurately detecting vegetation change, given the region's forest remnants are small and fragmented. Two Landsat TM scenes cover the QF (217 064 and 218 064) and near-date images were acquired for both from 1990, 2000, 2004 and 2010. Acquiring near-date images minimized differences in sun elevation angle and shadowing effects. Where possible, images were also chosen from July (end of dry season) to enhance spectral differences between grassy and woody vegetation and to minimize cloud occurrence (all images had <5% cloud cover). Images were downloaded from USGS, which were level 1T processed (orthorectified) and projected to UTM Zone 23S.

Acquired images were converted to reflectance values (using the published post-launch gain and offset values; NASA Goddard Space Flight Center 2011) and atmospheric effects were corrected using the QUAC method available in the classification software ENVI (ENVI 2010). Images were then combined using a geographical mosaic and clipped to the QF boundary, which was defined by intersecting a map of local municipalities (IBGE 2005) with the region officially defined as the QF by CODEMIG (2010).

A supervised, pixel-based classification algorithm was used to classify land cover classes. This approach was used over object-based methods because the later has been found to suffer from absorption of small rare classes (such as mining) into larger objects (Robertson & King 2011). The 'baseline' (2010) image was classified into six land cover classes (forest, grass, mining, plantations, urban and water). For each class, training pixels were selected based on field knowledge and higher-resolution Quickbird imagery. Spectral information was extracted from bands 1–7 (excluding the panchromatic band 6) and two vegetation indices: NDVI and Tasseled Cap (Jensen 2005). Importantly, significant separation in spectral signatures was found between training pixels from each land cover class. Both Jeffries-Matursita and Transformed Divergence separability statistics (Richards 1999) were >1.9 for all comparisons, indicating that between-class variation was significantly greater than within-class variation. The Spectral Angle Mapper (SAM) technique was then used to classify the baseline image in ENVI (ENVI 2010).

Pre-baseline images were initially processed using an image differencing and thresholding approach to identify pixels that had undergone a change in land cover (Mas 1999). This technique produced a 'change image' by subtracting a date-1 band (NDVI was used) from the corresponding date-2 band. A threshold value was then applied to produce a binary mask of 'change' and 'no change' (Figure 2). The threshold value was set to the 5% upper and lower histogram values of NDVI difference, therefore the absolute threshold value differed for each time step (i.e. 1990-2000 vis. 2000-2004). The binary image of change pixels was then overlaid with the date-2 image and only these pixels were classified. The advantage of using this method is that it reduces the number of pixels to be classified, which may also reduce omission and commission errors; however, accuracy depends on the threshold's ability to detect changes between land cover classes. To classify change pixels, spectral signatures collected from the 2010 baseline image were used and classification was performed as described previously. The advantage of utilizing 2010 spectral information was that it reduced the effort required to re-train classes (a task which was not possible for the 2000 and 2004 images since ground truth information was not available); however, the accuracy will depend on the temporal stability of spectral signatures for all land cover classes.

The time series of land cover classes were then converted into land use classes using a combination of cartographic information and time series decision rules. For the land cover class of grass, a native vegetation map (SEMAD 2010) was used to distinguish between native grasses (vegetation of Campo, Canga and Cerrado) and non-native grassy fields, which were primarily low-density or abandoned cattle grazing properties. The time series rules were used to correct changes in land cover that were not changes in land use. The time series rules were as follows: 'plantation to grass' was reclassified to stable plantation, since this land cover transition reflected plantation harvest, rather than plantation abandonment; 'plantation to forest' was reclassified as stable plantation, since this was considered an unrealistic transition; 'urban to non-urban' was reclassified to stable urban, also an unrealistic transition; 'field to forest to field' was reclassified as stable field, since it was assumed regrowth occurred only if forests persisted; and 'mining to grassy' was reclassified as stable mining, since rehabilitated land remained in use by mining companies.

To assess accuracy, a crisp (one class per pixel) pixel-based assessment was used to collect spectra and a stratified random sampling protocol was used to select ground truth points ensuring that rare classes were sampled (Foody 2011; Stehman 2009). Sample locations were generated using ENVI, reference (or ‘ground truth’) information on land use was collected from higher-resolution imagery for these points, and confusion matrices were generated to illustrate omission and commission errors and thus producer’s and user’s accuracy. Three accuracy assessments were performed. First, the 2010 land use classification was assessed against high-resolution Quickbird imagery from 2010 to determine the accuracy of the supervised classification. Accuracy was above 90% for all land use classes (Table 3). Second, the image differencing and thresholding approach was assessed to determine its ability to detect change. Comparisons were made between the ‘change’/‘no change’ mask and a combination of the 2010 Quickbird imagery and a 1990 orthorectified digital photograph. Results showed that change was accurately detected (Table 4); however, 63% of pixels detected as change actually underwent no change, suggesting a higher change detection threshold may have been useful for some land use classes, however it was considered more appropriate to overestimate potential change pixels, rather than risk not detecting them at all (Table 4). Third, the accuracy of using 2010 spectral information to classify pre-baseline land use maps was assessed by comparing the 1990 land use change map with the 1990 orthorectified photograph (Table 5). Accuracy was above 90% for all land use classes, indicating that errors in Table 4 were corrected for through classification. In this study, quantitative field data accuracy assessment was not possible; however, each land use class of interest here was detectable from high resolution images.

2.3 Comparisons with other studies

To compare the processes of land use change within the QF with those found in nearby non-mining regions, we collated a series of published case studies. Comparisons were limited (by availability) to five regional studies (Becker et al. 2004; Castanheira 2010; Freitas et al. 2010; Lira et al. 2012; Teixeira et al. 2009) plus one biome wide analysis (Calmon et al. 2011) and one State-wide analysis (SEMAD 2010). We compared these studies with QF results by evaluating similarities and differences in regional land use composition and extent, land use transitions and land use transition rates.

3. RESULTS AND DISCUSSION

3.1 QF land use change

In 2010, the QF mining region was composed of a mosaic of land uses interspersed throughout highly fragmented forests and native grasslands. The land use map showed that less than half the region's native vegetation remained and the majority of land was used for some form of production (Figure 1). Low-density or abandoned cattle-grazing pastures (classified here as fields) were dominant, followed by *Eucalyptus* plantations, urban areas and mining operations (Table 1). The regional extent of land use classes changed over time as the result of dynamic land use transitions (Table 1).

Eight land use transitions were observed between 1990 and 2010 (Table 2). Native vegetation was cleared for multiple land uses, including fields, mines and urban. Land use transitions also occurred between land use classes, i.e. fields were transitioned for both plantation expansion and urban development. A small amount of native forest regrowth took place; however, the rate of forest regrowth steeply declined over time (Table 2). No evidence of revegetation or rehabilitation of mine sites with forest cover was evident from the spatial data between 1990 and 2010.

In addition to these proximate causes of land use change, we found mining operations indirectly influenced adjacent land users. These findings have been reported elsewhere (see Sonter et al. 2013b; Sonter et al. in review) and include: increased plantation expansion for charcoal production for use in pig iron and steel making (half the pig iron in Minas Gerais is produced using charcoal, of which 65% is produced from plantation forests; AMS 2012; IBGE, 2012); an increase in offsite (beyond mine lease) deforestation rates potentially driven by competition between mining companies and urban developers; and a decline regional forest regrowth rates driven by increased regional charcoal production. These results illustrate the physical 'reach' of mining operations in this region, which were visible through a Land Change analysis in the QF mining region.

3.2 QF vs. surrounding non-mining landscapes

Previous studies undertaken within the Atlantic Forest illustrate landscapes that are highly altered and contain a mosaic of land uses, which was similar our findings in the QF. For example, the dominant land uses (forests and fields) and their relative proportions in the QF (fields dominated and native vegetation fell below 50% of the landscape; Figure 1) were consistent with other studies undertaken in nearby watersheds (Castanheira 2010), within the State of Minas Gerais (SEMAD 2010) and elsewhere in the Atlantic Forest biome (Becker et al. 2004; Lira et al. 2012; Teixeira et al. 2009). Other similarities included 1) the transition of forests to fields being the most extensive transition and 2) a steady increase in the extent of urbanized land over the past two decades (Lira et al. 2012).

Differences were associated with deforestation and regrowth trajectories, the occurrence of mining land use and the rate at which land used for plantation forestry expanded. Other studies reported that deforestation rates have slowed over time and forest regrowth rates (i.e. the transition of fields to forests) have increased, ultimately leading to a net increase in forest cover (Lira et al. 2012). In these non-mining landscapes an observed net increase in forest cover was explained by a combination of factors, including 1) increased enforcement of forest management legislation, specifically the Forest Code (Calmon et al. 2011) and 2) the combined effect of increasing land rents and modern agricultural practices, which have driven land abandonment and forest regrowth (Becker et al. 2004; Lira et al. 2012). Neither characteristic, however, was evident in the QF.

In the QF, deforestation rates have not declined since 2000 (Table 1, Table 2) and the influence of forest legislation on reducing deforestation rates appears to have been minimal during this time. While the rate of deforestation remained relatively stable over the past decade (increased enforcement may have at least prevented increased deforestation rates), the region’s second most important proximate cause of forest loss—i.e. mining—increased.

Under the Forest Code, which is Brazil’s national forest management policy, mining companies are permitted to clear forests, so long as they obtain an environmental license and compensate (or offset) for forest loss. Compensation involves activities such as revegetation and conservation and should result in ‘no-net-loss’ to forests in the region. The influence of offset projects on slowing regional deforestation, however, appears to be relatively insignificant as a result of poorly designed offsetting requirements (Sonter et al. 2014). Alternatively, large tracts of forested land surrounding mining operations are owned and inadvertently conserved by mining companies within the region since adjacent land users are excluded from development in these areas (Figure 1; Sonter et al. 2013b). These findings suggest that the operation of the mining industry has a significant influence on the processes of deforestation within the QF region, both as an observed cause of deforestation and as a potential source of conservation.

An increased rate of forest regrowth as a result of land abandonment was also not observed in the QF (Table 2). This was because a highly profitable, alternative land use option (plantation forestry) was available. Plantation forestry operations rapidly expanded in the QF (Table 1) and this was uncharacteristic of other Atlantic Forest landscapes. In the QF, plantations produce both cellulose (for paper production) and biomass for charcoal production. Charcoal production is used in part for domestic purposes and in part for steel making (driven by the mining of iron ore and global demands; Sonter et al. in review). This suggests that in addition to being a major proximate cause of deforestation, the operation of the mining industry also plays an important underlying role in driving plantation expansion in the QF, which was not evident in surrounding non-mining Atlantic Forest landscapes.

3.3 General processes of land use change in mining regions

While some similarities were found between the QF and surrounding non-mining landscapes, many differences were evident. Specifically, the QF contained additional highly profitably land uses, including mining and plantation forestry (Figure 1), which were driven by globalized markets for mineral resources (Sonter et al. in review). This result suggests mining regions undergo processes of land use change that are distinct from what may have been expected in absence of high quality mineral deposits and, as such, they should be managed differently. Knowing how to do this requires a general understanding of the processes of land use change that occur in these regions. From our results, it was possible to identify four potential generalizations: 1) the direct footprint of mining expands over time, 2) the offsite footprint of mining is extensive and also often expanding, 3) the direct and indirect use of land by mining causes environmental and social impacts, some of which are not captured by current management approaches, and 4) the footprints of mining and their associated impacts are driven by global factors, many of which are uncontrollable by local land holders. In this section we expand on each of these generalizations, drawing on published evidence from other mining regions to illustrate their generality and their implications for land management.

3.3.1 The direct footprint of mining expands over time

The aggregated land area used directly for mining in the QF expanded over time (Table 1; Figure 2), at a rate that also increased non-linearly (Table 2). The expansion of mining operations is common in many mining regions (for example, see the Pilbara in Australia and Rustenburg in South Africa; InfoMine 2012) and can be explained by four related factors. 1) The demand for minerals has grown across many mineral commodities (UNEP 2011). 2) During recent decades there has been a shift from underground mining to massive-scale surface mining operations due to 'economies of scale' (Prior et al. 2012). 3) Lower grades in metals (requiring more rock to be mined) and deeper viable coal deposits (requiring higher strip ratios for extraction) result in more land being required to produce the same amount of product (Mudd 2010). 4) As a result of the previous factors, the extent of tailings storage facilities and waste rock dumps has also grown (Franks et al. 2011).

Exploration activities are also increasing in scale. For example, Brazil increased investment in mineral exploration from USD234 million in 2009 to USD321 million in 2010 (Gurmendi 2011) and many mining regions are almost completely occupied with mineral exploration leases (USGS 2009). The spatial distribution of mineral exploration at the global scale is also changing, where a shift from 'green fields' to 'brown fields' is underway (ABS 2013). This shift results in the development of new mines within already established mining regions, rather than discovery and development of new mining regions. This trend is driven by higher probability of success in finding economically feasible reserves close to already established mining operations and the lower costs to exploit these reserves if found.

Expanding the direct footprint of mining and exploration within already established mining regions is expected to continue while their economically feasible mineral deposits remain. ‘Densification’ of mining regions is likely to elevate pressures on land, causing competition and conflict between mining and non-mining land users. Such should be expected to occur especially in regions already highly allocated for other non-mining forms of land use. Moran and Brereton (2013) illustrated this effect through the relationship between aggregate community complaints information and visual amenity over time in the Upper Hunter Valley in NSW Australia. Of course, when mineral resources are depleted, mine expansion will slow and exploration will cease. Following this, the direct footprint of mining will depend on regional land rehabilitation requirements and the success of these activities. It is worth noting, however, that once mineral resources are depleted the term ‘mining region’, as initially defined, no longer applies, although evidence of a ‘closed’ mining region without permanent impacts is yet to be demonstrated or predicted with certainty in planning.

3.3.2 The offsite footprint of mining is extensive and also often expanding

In the QF, the land used by mining companies extended beyond their onsite operations. Offsite footprints have previously been referred to as ‘shadow effects’ or ‘spill-over effects’ (Marshall 1982; Schueler et al. 2011) and these also appear common in mining regions. Specifically in the QF, we found land was used offsite for plantation forestry to produce charcoal to enable iron ore processing and steel making (Sonter et al. in review). In other iron mining regions, plantation charcoal production is also attracting attention in the context of climate change mitigation (Weldegiorgis & Franks 2013). Mining infrastructure also often has an offsite footprint. Transportation infrastructure (both for products and workforce, including rail and road), mineral processing, pelletizing and metal refining plants all increase in size with the direct footprint of mining (ABS 2010). For example, in the Bowen Basin coal region of Australia increased traffic volume transporting mining needs and a growing workforce has resulted in the need for significant upgrade of the region's road network. Further west in the same state, the planning for new railway corridors for the development of the Galilee Basin for coal production has been the subject of significant approvals and community conflict. On the other hand, the World Bank is examining the design of railway development for mining in East Africa to also create regional-scale synergies for opening land for food production by creating the ability to transport all commodities to markets.

The general trend is that the full extent of land used to support mining operations is in addition to its direct footprint (Sonter et al. 2013b), suggesting that regional management of mining should also consider land used offsite and the effects of this on adjacent land users. This is especially true since important feedbacks often exist between increasing offsite footprints and future mine expansion. For example, upgrading regional infrastructure increases the productive capacity of a region (Gurmendi 2011), thus providing new (and often cheaper) opportunities to expand onsite operations. Therefore the approval and management of offsite land use should be done considering its potential to catalyze future mine expansion and land use change within the region. The significance of this effect can be seen in the consideration by oil and gas companies to shift to offshore floating natural gas liquefaction plants to avoid the complexities of on-shore developments. Another example is the dynamic causal relationship that often occurs between mine expansion and urbanization in mining regions (Petkova-Rimmer et al. 2009; Roberts 1992). Finally, it is also possible for offsite footprints to extend beyond the mining region itself, making them uncontrollable through regional planning. An example of such an effect is where constraints to stockpiling in the source region result in the creating of stockpiles footprints in other locations, e.g., at distant ports located offshore. This point illustrates that our current definition of a mining region and their management, does not explicitly capture offsite footprints of mining.

3.3.3 Environmental and social impacts are caused by the footprints of mining

Land use change associated with the direct and offsite footprints of mining causes environmental and social impacts. Directly, surface mining operations displace soil, clear vegetation (e.g. Figure 2), reconfigure natural landscapes and alter ecosystem function and services (Simmons et al. 2008), they can cause both enhancement and loss of regional livelihoods and of quality of life depending on local circumstances (Moran et al. 2013; Schueler et al. 2011). Impacts caused by the direct footprint of mining have received significant attention in the literature and they are the subject of impact assessments and licensing conditions, within which mining operations are required to avoid, minimize rehabilitate or offset these impacts; although their success in doing so is debatable (Sonter et al. 2014). However, the impacts caused by offsite footprints have received considerably less attention and, we suggest here, that these are not currently captured by impact assessments or licensing conditions, although some evidence from regional and strategic impact assessments suggests these tools can be used to capture these impacts. The challenges in managing these impacts are associated with assigning responsibility, since a direct link between cause and impacts cannot always be easily established; however, overlooking them will have significant implications for achieving sustainable development goals both within and beyond mining regions.

3.3.4 Land use change in mining regions is driven by global factors

Mineral resources are traded within globalized markets (Barbier 2000; Bridge 2004) and as a result there is often great distance between the drivers of land use change (the demand for products, particularly for minerals from developing Asian countries) and the environmental and social impacts that occur locally (Fearnside et al. 2013; Lambin et al. 2001). This was observed in the QF, where the global demand for iron and steel caused a transformation of the region to produce iron ore and charcoal (Figure 1; Sonter et al. in review). The challenge in managing these processes of land use change, then, is also linked to managing demand for mineral resources elsewhere. While this task is difficult—since the local land holders experiencing impacts have little to no control over global drivers—failing to do so will limit the effectiveness of long-term regional planning, especially if a 'business as usual' demand scenario is incorrectly assumed. The growing realization of the significance of this effect is resulting in many countries introducing forms of 'royalties to regions' policies, which preferentially direct revenues and taxes from mineral and energy commodity exploitation to the local region to deal with environmental and community consequences of being the source location of mining activity. This is a governance response to the phenomenon that physical needs for sustainable development in one place do not create an inability for equitable development in another location, for example, the supply of copper from Peruvian Andean communities to rapidly urbanizing China. More broadly, the influence of 'teleconnections' (i.e. the distant link between drivers of change and their impacts) is increasing throughout the world in other non-mining regions (Liu et al. 2013), for example in regions producing biofuels (Reenberg & Fenger 2011). Therefore more general lessons on managing global drivers of land use change could be learned from the world's mining regions, where global mineral markets have been in effect for decades.

4. CONCLUSION

Our results suggest that the processes of land use change in mining regions are distinguishable from those occurring elsewhere. This finding suggests land management approaches should be specifically tailored for mining regions. We propose four generalizations regarding the observed processes of land use change, which could be used to guide policy development and land management in existing and emerging mining regions throughout the world. Future research could test the validity of our generalizations by analyzing the processes of land use change in other mining regions, at different stages of resource development. Questions raised herein point to the need for a more thorough examination of the definition and scope of mining (or resource) regions and research into the processes operating within them and impacts at distance from them. To do this, our results suggest a spatially and temporally explicit Land Change analysis coupled with remote sensing information is likely to be useful in many cases and essential in some.

ACKKNOWLEDGEMENTS

We thank Magnus Wettle and Birte Schoettker for remote sensing advice and two anonymous reviewers for helpful feedback on a previous version. Britaldo Soares-Filho received financial support from CNPQ, FAPEMIG and CLUA.

REFERENCES

ABS. 2010. Mining Capital Expenditure in Australia September 2010. Australian Bureau of Statistics.

ABS. 2013. Mineral and Petroleum Exploration, Australia June 2013. Australian Bureau of Statistics.

AMS. 2012. Demand and Availability. Associação Mineira da Silvicultura. Belo Horizonte, Brazil.

Barbier, E. B. 2000. Links between economic liberalization and rural resource degradation in the developing regions. Agricultural Economics 23:299-310.

Becker, F. G., G. V. Irgang, H. Hasenack, F. S. Vilella, and N. F. Verani. 2004. Land cover and conservation state of a region in the southern limit of the atlantic forest. Brazilian Journal of Biology 64:569-582.

Bridge, G. 2004. Contested terrain: Mining and the environment. Annual Review of Environment and Resources 29:205-259.

Calmon, M., P. H. S. Brancalion, A. Paese, J. Aronson, P. Castro, S. C. da Silva, and R. R. Rodrigues. 2011. Emerging Threats and Opportunities for Large-Scale Ecological Restoration in the Atlantic Forest of Brazil. Restoration Ecology 19:154-158.

Castanheira, L. A. 2010. Estudo das Mudanças de Uso e Cobertura da Terra no Parque Nacional da Serra do Cipó e Entorno no Período de 1989 a 1999. Page 147. Universidade Federal de Minas Gerais, Belo Horizonte, Brazil.

CODEMIG. 2010. Geological Mapping of Minas Gerais. Companhia de Desenvolvimento Economico de Minas Gerais (CODEMIG).

DNPM. 2012. Processos Minerarios. Sistema de Informações Geográficas da Mineração (SIGMINE). Departamento Nacional de Produção Mineral (DNPM),, Brazil.

ENVI. 2010. ENVI User's Guide. Exelis Visual Information Solutions, Boulder, Colorado.

Fearnside, P. M., A. M. R. Figueiredo, and S. C. M. Bonjour. 2013. Amazonian forest loss and the long reach of China’s influence. Environment, Development and Sustainability 15:325-338.

Foody, G. M. 2011. Classification Accuracy Assessment. IEEE Geoscience and Remote Sensing Society Newsletter:8-14.

Franks, D. M., D. V. Boger, C. M. Côte, and D. R. Mulligan. 2011. Sustainable development principles for the disposal of mining and mineral processing wastes. Resources Policy 36:114-122.

Franks, D. M., D. Brereton, and C. J. Moran. 2013. The cumulative dimensions of impacts in resource regions. Resources Policy.

Freitas, S. R., T. J., Hawbaker, and J. P Metzger. 2010. Effects of roads, topography, and land use on forest cover dynamics in the Brazilian Atlantic Forest. Forest Ecology and Management. 259:410-417.

Gurmendi, A. C. 2011. The Mineral Industry of Brazil. U.S. Geological Survey.

Hammond, D. S., V. Gond, B. de Thoisy, P. M. Forget, and B. P. E. DeDijn. 2007. Causes and consequences of a tropical forest gold rush in the Guiana Shield, South America. Ambio 36:661-670.

IBGE. 2005. Digital Municipal Mesh. Instituto Brasileiro de Geografia e Estatıstica (IBGE). Brazil.

IBGE. 2012. Extraction of Plant Production and Forestry. Instituto Brasileiro de Geografia e Estatística (IBGE). Brazil

InfoMine. 2012. Company & Property Mine. InfoMine.

Irons, J. R., H. Lachowski, and C. Peterson. 1980. Remote sensing of surface mines- A comparative study of sensor systems. Pages 1041-1053. International Symposium on Remote Sensing of Environment, San Jose, Costa Rica.

Irons, J. R., H. Lachowski, and C. Peterson. 1986. Potential utility of the Tematic Mapper for surface mine monitoring. Photogrammetric Engineering and Remote Sensing 52:389-396.

Jacobi, C. M., and F. F. do Carmo. 2008. The contribution of ironstone outcrops to plant diversity in the Iron Quadrangle, a threatened Brazilian landscape. Ambio 37:324-326.

Jensen, J. R. 2005. Introductory digital image processing: a remote sensing perspective. Upper Saddle River, Prentice Hall.

Lambin, E. F., and M. Linderman. 2006. Time series of remote sensing data for land change science. Ieee Transactions on Geoscience and Remote Sensing 44:1926-1928.

Lambin, E. F., B. L. Turner, H. J. Geist, S. B. Agbola, A. Angelsen, J. W. Bruce, O. T. Coomes, R. Dirzo, G. Fischer, C. Folke, P. S. George, K. Homewood, J. Imbernon, R. Leemans, X. B. Li, E. F. Moran, M. Mortimore, P. S. Ramakrishnan, J. F. Richards, H. Skanes, W. Steffen, G. D. Stone, U. Svedin, T. A. Veldkamp, C. Vogel, and J. C. Xu. 2001. The causes of land-use and land-cover change: moving beyond the myths. Global Environmental Change-Human and Policy Dimensions 11:261-269.

Lira, P. K., L. R. Tambosi, R. M. Ewers, and J. P. Metzger. 2012. Land-use and land-cover change in Atlantic Forest landscapes. Forest Ecology and Management 278:80-89.

Liu, J. G., V. Hull, M. Batistella, R. DeFries, T. Dietz, F. Fu, T. W. Hertel, R. C. Izaurralde, E. F. Lambin, S. X. Li, L. A. Martinelli, W. J. McConnell, E. F. Moran, R. Naylor, Z. Y. Ouyang, K. R. Polenske, A. Reenberg, G. D. Rocha, C. S. Simmons, P. H. Verburg, P. M. Vitousek, F. S. Zhang, and C. Q. Zhu. 2013. Framing Sustainability in a Telecoupled World. Ecology and Society 18.

Marshall, I. B. 1982. Mining, land use and the environment. Lands Directorate, Environment Canada, Ottawa.

Mas, J. F. 1999. Monitoring land-cover changes: a comparison of change detection techniques. International Journal of Remote Sensing 20:139-152.

Matschullat, J., R. P. Borba, E. Deschamps, B. R. Figueiredo, T. Gabrio, and M. Schwenk. 2000. Human and environmental contamination in the Iron Quadrangle, Brazil. Applied Geochemistry 15:181-190.

MME. 2011. Plano Nacional de Mineracao 2030: Geologia, Mineracao e Transformacao Mineral. Ministério de Minas e Energia (MME), Brasilia.

Moran, C. J., and D. Brereton. 2013. The use of aggregate complaints data as an indicator of cumulative social impacts of mining: A case study from the Hunter Valley, NSW, Australia. Resources Policy.

Moran, C. J., D. M. Franks, and L. J. Sonter. 2013. Using the multiple capitals framework to connect indicators of regional cumulative impacts of mining and pastoralism in the Murray Darling Basin, Australia. Resources Policy 38:733-744.

Mudd, G. M. 2010. The Environmental sustainability of mining in Australia: key mega-trends and looming constraints. Resources Policy 35:98-115.

NASA Goddard Space Flight Center. 2011. Landsat 7 Science Data Users Handbook.

Petkova-Rimmer, V., S. Lockie, J. Rolfe, and G. Ivanova. 2009. Mining developments and social impacts on communities: Bown Basin case studies. Rural Society 19:211-228.

Prior, T., D. Giurco, G. Mudd, L. Mason, and J. Behrisch. 2012. Resource depletion, peak minerals and the implications for sustainable resource management. Global Environmental Change-Human and Policy Dimensions 22:577-587.

Reenberg, A., and N. A. Fenger. 2011. Globalizing land use transitions: the soybean acceleration. Geografisk Tidsskrift-Danish Journal of Geography 111:85-92.

Reid, R. S., T. P. Tomich, J. Xu, H. Geist, A. Mather, R. S. DeFries, J. Liu, D. Alves, B. Agbola, E. F. Lambin, A. Chabbra, T. Veldkamp, K. Kok, M. van Noordwijk, D. Thomas, C. Palm, and P. H. Verburg. 2006. Linking Land-Change Science and Policy: Current Lessons and Future Integration. Pages 151-171 in E. Lambin, and H. J. Geist, editors. Land Use and Land Cover Change: Local Processes and Global Impacts. Springer-Verlag Berlin.

Ribeiro, M. C., J. P. Metzger, A. C. Martensen, F. J. Ponzoni, and M. M. Hirota. 2009. The Brazilian Atlantic Forest: How much is left, and how is the remaining forest distributed? Implications for conservation. Biological Conservation 142:1141-1153.

Richards, J. A. 1999. Remote Sensing Digital Image Analysis. Springer-Verlag, Berlin.

Roberts, J. T. 1992. SQUATTERS AND URBAN-GROWTH IN AMAZONIA. Geographical Review 82:441-457.

Robertson, L. D., and D. J. King. 2011. Comparison of pixel- and object-based classification in land cover change mapping. International Journal of Remote Sensing 32:1505-1529.

Schueler, V., T. Kuemmerle, and H. Schroder. 2011. Impacts of Surface Gold Mining on Land Use Systems in Western Ghana. Ambio 40:528-539.

SEMAD. 2010. Mapeamento da Cobertura Vegetal. Secretaria de Estado de Meio Ambiente e Desenvolvimento Sustentável (SEMAD) Minas Gerais, Brazil.

Simmons, J. A., W. S. Currie, K. N. Eshleman, K. Kuers, S. Monteleone, T. L. Negley, B. R. Pohlad, and C. L. Thomas. 2008. Forest to reclaimed mine land use change leads to altered ecosystem structure and function. Ecological Applications 18:104-118.

Sonter, L. J., C. J. Moran, and D. J. Barrett. 2013a. Modeling the impact of revegetation on regional water quality: a collective approach to manage the cumulative impacts of mining in the Bowen Basin, Australia. Resources Policy 38:670-677.

Sonter, L. J., D. J. Barrett, C. J. Moran, and B. S. Soares-Filho. 2013b. A Land System Science meta-analysis suggests we underestimate ‘intensive’ land uses. Journal of Land Use Science.

Sonter, L. J., D. J. Barrett, and B. S. Soares-Filho. 2014. Offsetting the impacts of mining to achieve no-net-loss to native vegetation and its biodiversity. Conservation Biology. In press.

Sonter, L. J., D. J. Barrett, B. S. Soares-Filho, and C. J. Moran. in review. The global demand for steel drives extensive land use change in Brazil. Global Environmental Change-Human and Policy Dimensions.

Stehman, S. V. 2009. Sampling designs for accuracy assessment of land cover. International Journal of Remote Sensing 30:5243-5272.

Teixeira, A. M. G., B. S. Soares, S. R. Freitas, and J. P. Metzger. 2009. Modeling landscape dynamics in an Atlantic Rainforest region: Implications for conservation. Forest Ecology and Management 257:1219-1230.

Townsend, P. A., D. P. Helmers, C. C. Kingdon, B. E. McNeil, K. M. de Beurs, and K. N. Eshleman. 2009. Changes in the extent of surface mining and reclamation in the Central Appalachians detected using a 1976-2006 Landsat time series. Remote Sensing of Environment 113:62-72.

Turner, B. L., E. F. Lambin, and A. Reenberg. 2007. The emergence of land change science for global environmental change and sustainability. Proceedings of the National Academy of Sciences of the United States of America 104:20666-20671.

UNEP. 2011. Decoupling natural resource use and environmental impacts from economic growth in M. Fischer-Kowalski, M. Swilling, E. U. von Weizsacker, Y. Ren, Y. Moriguchi, W. Crane, F. Krausmann, N. Eisenmenger, S. Giljum, P. Hennicke, P. Romero Lankao, and A. Siriban Manalang, editors. Report of the Working Group on Decoupling to the International Resource Panel.

USGS. 2009. Minerals Yearbook, Brazil. U.S. Geological Survey.

Viana, M. B., and M. A. A. Bursztyn. 2010. Environmental regularization of mining activities in the State of Minas Gerais, Brazil. Rem-Revista Escola De Minas 63:363-369.

Weldegiorgis, F. S., and D. M. Franks. in press. Social dimensions of energy supply alternatives in steelmaking: Comparison of biomass and coal production scenarios in Australia. Journal of Cleaner Production.

FIGURE CAPTIONS

Figure 1: Quadrilátero Ferrífero mining region. Inset top left inset illustrates the location of the QF within Brazil and Minas Gerais. The main figure shows the 2010 land use classification map.

Figure 2: Land cover classification procedure, showing a sub-section (see inset) of the QF mining region.

TABLES

Table 1: Land use classes over time

	Area (100 ha)			
Land use	***1990***	***2000***	***2004***	***2010***
Forest	921	889	879	858
Grass	71	70	69	68
Fields	868	879	878	869
Plantations	69	80	88	110
Urban	43	51	53	57
Mine	8	11	13	18
Total				***1980***

Table 2: Annual rate of land use transitions

		Annual rate (ha.yr^{-1})		
From	**To**	**1990-2000**	**2000-2004**	**2004-2010**
Forest	Field	3019	2098	2363
	Urban	144	59	103
	Mining	178	203	432
Grass	Urban	11	25	40
	Mining	88	146	136
	Plantation	2	4	53
Field	Urban	652	423	504
	Mining	83	102	92
	Forest	629	281	258
	Plantation	586	1245	2587

Table 3: Accuracy assessment of the baseline image (2010) and 1990 land cover

		Quickbird (Observed)							**Accuracy (%)**	
		Mine	**Urban**	**Grass**	**Plantation**	**Forest**	**Water**	**Total**	**Producer's**	**User's**
2010 Classification (Predicted)	**Mine**	85	2	7	0	0	0	94	**100.00**	**90.43**
	Urban	0	88	4	0	1	1	94	**93.12**	**93.62**
	Grass		11	189				200	**90.80**	**94.50**
	Plantation	0	0	0	88	8	0	96	**96.70**	**91.67**
	Forest	0	0	5	3	190	0	198	**95.48**	**95.96**
	Water	0	0	0	0	0	50	50	**98.03**	**100.00**
	Total (n+i)	85	101	205	91	199	51	732		

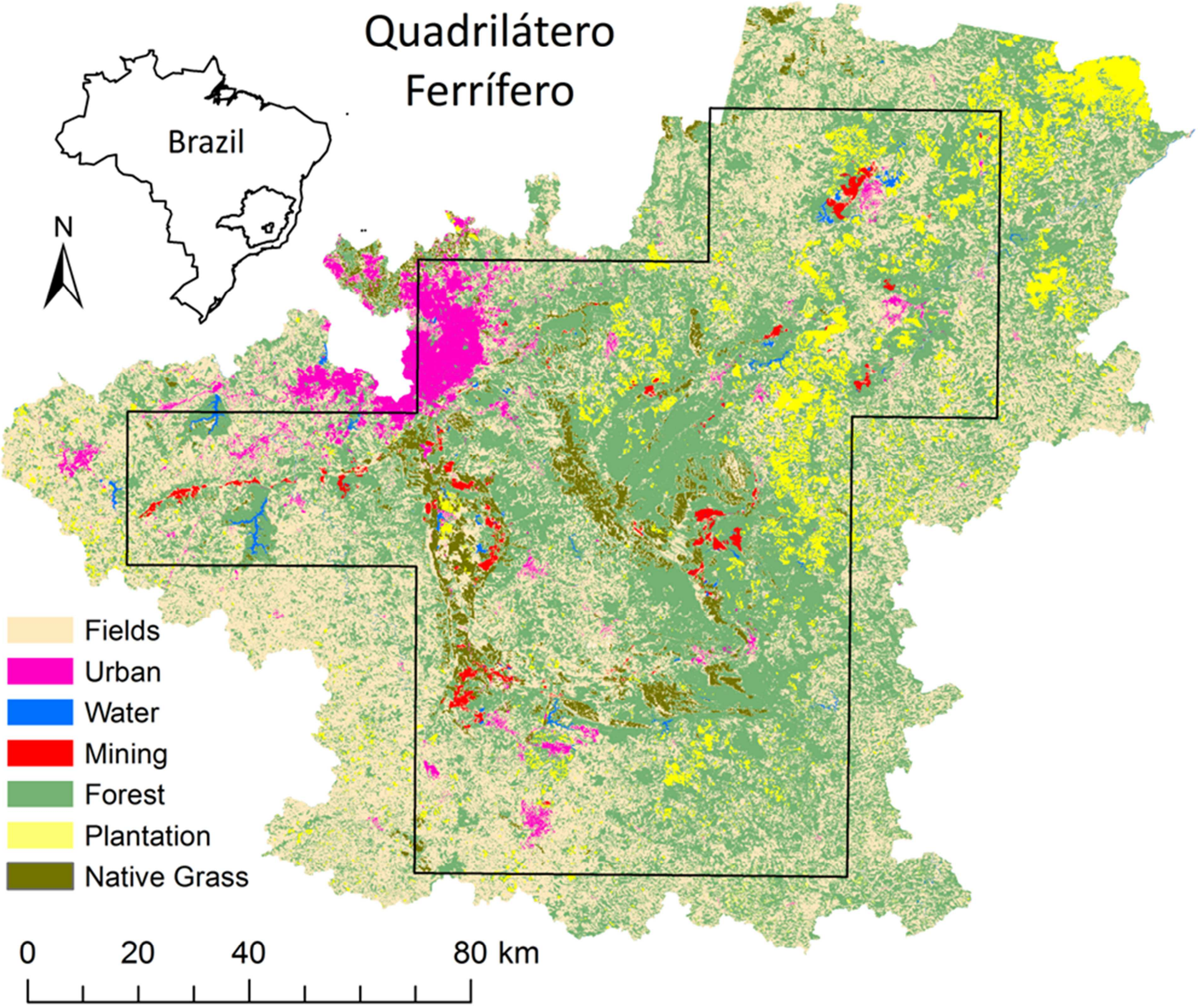
Table 4: Accuracy assessment of 1990 change mask

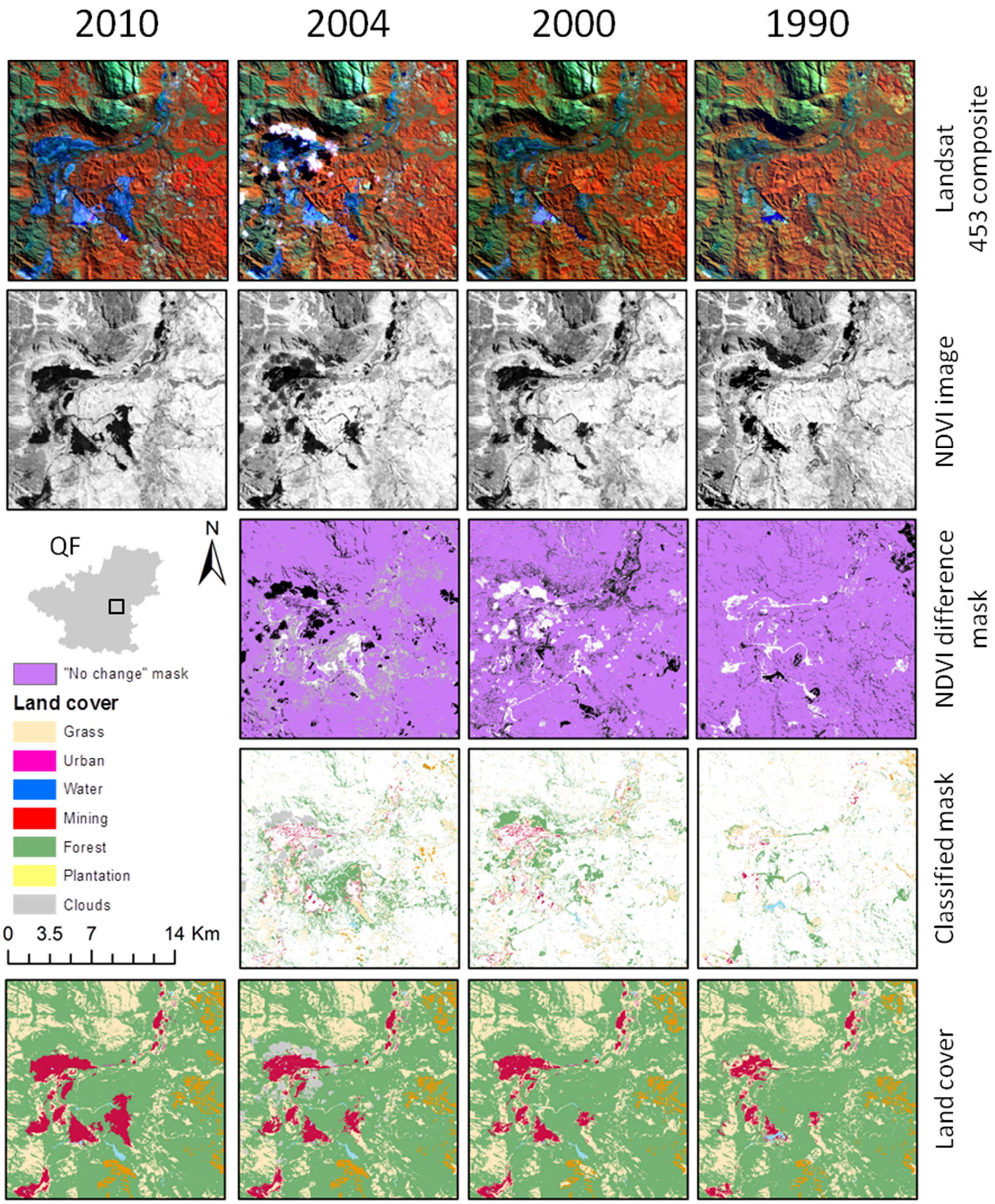
		Quickbird & Ortho-photo (Observed)			**Accuracy (%)**	
		CHANGE	**NO CHANGE**	**Total**	**Producer's**	**User's**
1990 Change Mask (Predicted)	**CHANGE**	125	72	197	**100.00**	**63.45**
	NO CHANGE	0	200	200	**73.52**	**100.00**
	Total (n+i)	125	272	397		

1 Table 5: Accuracy assessment of 1990 land cover within change mask

		Ortho-photo (Observed)							Accuracy (%)	
		Mine	**Urban**	**Grass**	**Plantation**	**Forest**	**Water**	**Total**	**Producer's**	**User's**
1990 Classification (Predicted)	**Grass**	0	2	187	0	8	0	197	**96.39**	**94.92**
	Forest	0	0	7	1	192	0	200	**95.47**	**96.00**
	Total (n+i)	0	2	194	1	200	0	397		

2


Quadrilátero
Ferrífero
Brazil
N
Fields
Urban
Water
Mining
Forest
Plantation
Native Grass
0
20
40
80 km


2010
2004
2000
1990
Landsat
453 composite
NDVI image
NDVI difference
mask
Classified mask
Land cover
QF
N
"No change" mask
Land cover
Grass
Urban
Water
Mining
Forest
Plantation
Clouds
0
3.5
7
14 Km

Highlights

- Mining regions undergo abrupt and extensive land use change (LUC)
- A Land Change analysis was used to investigate LUC in Brazil’s Iron Quadrangle (QF)
- Processes of LUC within the QF were distinct from those in non-mining regions
- Some similarities between the QF and other mining regions were also evident
- Four generalisations were identified to help guide land management in mining regions