INTRODUCTION

Being at the top of the food chain and found at low densities, the conservation of large carnivores is a challenge to biodiversity conservation (Chapron G. et al. 2014). With their requirement of large home ranges, carnivores occupy vast areas to harbor viable populations (Purvis et al. 2000). Since historical times, carnivores are in conflict with humans for food and resources, often resulting in their demise (Gittleman et al. 2001). Several factors, such as habitat destruction and excessive hunting by humans in lieu of real or perceived threat to people and their livestock, the use of body parts for traditional medicine, have extirpated many populations (Woodroffe 2000, Clark et al. 1996, Check 2006) while shrinking, fragmenting, and isolating most others to varying degrees (Crooks 2002, Proctor et al. 2005). Small and isolated such populations are prone to local extinctions (Wilcox and Murphy 1985, Woodroffe and Ginsberg 1998), therefore managing them in a meta-population framework (Harrison 1991, Hanski 1994) by connecting through habitat corridors (Beier and Noss 1998, Noss 1987) provides individuals an opportunity to disperse, establish residency and reproduce, reduces the overall risk of extinction (Purvis et al. 2000, Banerjee and Jhala 2012). In comparison to other large cats, tigers having relatively high K selected life history, dispersal and immigration play a vital role in long-term viability of tiger populations (Chapron et al. 2008). Small isolated tiger populations are likely to face extinction due to demographic stochasticity, inbreeding depression (Frankham et al 2002) and poaching (Chapron et al. 2008, Kenney et al. 1995). Therefore, habitat connectivity is essential in sustaining regional populations of tigers, as they require contiguous forest connectivity for dispersal and genetic exchange between populations (Smith 1993; Yumnam et. al. 2014).

The impacts of habitat fragmentation can be mitigated by preventing or reversing population isolation through connectivity conservation (Crooks and Sanjayan 2006), which facilitates the accommodation of species to the shifts in their natural domains due to climate change and maintain their evolutionary potential (Crooks and Sanjayan 2006,

Revilla and Wiegand, 2008; Lowe and Allendorf 2010). Wildlife conservation has emphasized on two primary strategies to promote connectivity. The first focuses on conserving areas that facilitate movement; while the second focuses on restoring connectivity across areas that impede movement. Most connectivity analyses have focused on the former strategy where modeling and mapping areas important for movement are important. In the majority of conservation studies, it is observed that a corridor is a continuous remnant of habitat between otherwise isolated habitat patches (Inglis and Underwood 1992; Beier and Noss 1998). Thus, a corridor is not merely an animal movement path, but also a conservation intervention (e.g., land protection, restoration, and management), when applied to a portion of the potential movement area between habitat patches, it achieves specific connectivity goals in landscapes that would otherwise be fragmented by urban, agricultural, or industrial land uses. The phrase "landscape permeability" which suggests (1) the importance of dynamic processes, (2) the species-specific nature of obstacles to movements, is often substituted for connectivity. It compels conservationists to think about the landscape (including the "matrix" of unprotected land) as a whole, rather than focusing on protected areas and narrow defined corridors (Forman and Godron 1986).

Studies on modeling connectivity has always focused on developing reliable linkage designs (Beier et al. 2008) whereas less attention has been paid to procedures for mapping regional connectivity within numerous natural landscape blocks over large areas, such as a nation, province, or eco-region. Among the methods and metrics used for analyzing connectivity, the two recent and complementary approaches that have gained popularity and provided important conceptual improvements for decision making in conservation planning are graphs (Urban and Keitt 2001) and habitat availability metrics (Pascual-Hortal and Saura 2006). With intensive efforts put towards advancing the best way to quantify and incorporate connectivity in landscape planning, much research work is carried out in field understanding the

underlined meta-population dynamics, focusing on patch based description of connectivity (Moilanen and Hanski 2001). Models distinguishing between habitats of varying quality for a species are prerequisites while estimating functional connectivity, as animals are believed to select movement paths in the same way they choose habitat (Beier et al. 2008). Therefore a reduced ecological cost (e.g., mortality risk) is accounted when individuals move through favourable habitats (Rayfield et al. 2010), demonstrating the significance of high quality habitats to be more permeable to movement than low quality habitats.

Conceived as a landscape feature (Tischendorf and Fahrig 2000) and defined as the connectedness of habitat for a particular species (Fischer and Lindenmayer 2007), connectivity can be a simple structural indices to more complex, biologically detailed, dynamic and spatially explicit meta-population models (Hanski and Ovaskainen 2000, Calabrese and Fagan 2004, Wiegand et al. 2005). Hence, measuring functional connectivity involves understanding the relation between structural characteristics of the landscape with ecological and behavioral characteristics of the species or community of species (Adriaensen et al. 2003). There are several approaches available to evaluate connectivity across complex landscapes, including least-cost path modeling (Adriaensen et al. 2003, Cushman et al. 2010), circuit theory (McRae and Beier, 2007; McRae et al., 2008), other forms of network analysis (Urban et al., 2009), resistant kernel modeling (Compton et al., 2007; Cushman et al., 2010b), agent-based movement (Palmer, Coulon and Travis, 2011), gene flow simulations (Landguth and Cushman, 2010), statistical modeling (Cushman et al., 2006; Compton et al., 2007; Spear et al., 2010) or empirically derived understandings from detailed movement data (Sawyer et al., 2009; Cushman et al., 2011).

In the light of conservation of Tiger, rigorous assessment of the influence of connectivity in maintaining meta-population framework is a necessary component in planning. The tiger (*Panthera tigris*) acts as a umbrella and focal species for the conservation of forested

ecosystems throughout its range in Asia (Tilson and Seal 1987). Although widely debated (Simberloff, 1998; Caro and O'Doherty, 1999; Lindenmayer and Fischer, 2003; Wiens et al., 2008), the focal species concept is important in landscape planning for wildlife management and conservation as it allows action with incomplete knowledge, address the conservation or management requirements of whole communities by focusing on a species subset (Lambeck, 1997). The approach is widely used not only to design protected areas (e.g. Noss et al., 1999; Gopal 2014) but also to plan wildlife linkages (i.e. corridor networks for multiple species), to maintain connectivity among protected areas (e.g. Sanderson et al., 2002b; Rouget et al., 2006; Thorne et al., 2006; Beier et al., 2008; Yumnam et. al. 2014).

Conserving the tiger typifies the prospects and challenges inherent in the current paradigm of fragmented small populations and landscape based conservation models in large carnivores (Dolrenry et al 2014). Extant tiger populations are confined to fewer than 7% of their historical range in patchily distributed habitats across a range of twelve regional tiger conservation landscapes (TCLs) in southern and north-eastern Asia (Dinerstein et al. 2007). Six global priority TCLs of long-term tiger conservation significance are present in the Indian subcontinent alone. These Indian TCLs are important for global tiger recovery as they harbor over 60% of the estimated global population of ~3,000 wild tigers (Sanderson et al. 2006, Jhala et al 2011), and >60% of the global genetic variation in the species (Mondol et al. 2009). The high genetic variation seen in Indian tigers could be attributed to historically high population sizes, numbering about 50,000 individuals until c. 200 years ago, and habitat contiguity that permitted genetic exchange between the various regional tiger populations in the area (Mondol et al. 2009). Due to change in land ownership and forest use policy in the mid nineteenth century during British rule and again during the early years of India's independence a century later, much of the forest was cleared for timber and agricultural needs (Rangarajan 1996, Rangarajan 2006). This change in land use combined with

organized trophy hunting and bounty driven extermination resulted in severe decline, fragmentation and isolation of tiger populations throughout India (Rangarajan 2006, Narain et al. 2005). The strong wildlife legislation (The Indian Wildlife (Protection) Act, 1972) and the launching of Project Tiger in 1973 where the tiger, the top charismatic carnivore was used as a flagship and umbrella species for conserving the biodiversity of India's forested ecosystems marked the beginning of a new conservation era in India. Currently, within the six tiger occupied landscapes of India (Jhala et al. 2008), habitat contiguity varies extensively, with the best being within the Western Ghats and the North East, while fragmentation is highest in the Shivalik-Gangetic Plain and the Central Indian Landscapes (Jhala et al. 2011; Yumnam et. al. 2014). Most of the connecting habitats in these landscapes are not within the legal domain of protected areas and are often lost to burgeoning development demands of a growing economy and attrition by human consumptive uses.

Currently in India, the once contiguous tiger population is now fragmented with source populations primarily restricted to tiger reserves. A tiger reserve is legally mandated to designate a critical core area wherein human habitation and resource extraction is not permitted (Wildlife Protection Act 1972, amendment 2005, (The Indian Wildlife (Protection) Act, 1972). This core is surrounded by a buffer zone, which is essentially a multiple use area, wherein conservation objectives are to be given precedence over other land uses. Breeding populations of tigers are mostly located in the core area of tiger reserves, while the buffers usually serve as population sinks (Karanth et al. 2005, Jhala et al. 2011a,b). The size of these tiger reserves vary between 344 km2 to 3,150 km2 (average 1,321 km2), with tiger densities ranging from about 0.1 to 20 individuals per 100 km2 (Jhala et al. 2011, Jhala et al. 2011a, Walston et al. 2010). For a demographically viable tiger population, a minimum of 20 to 25 breeding units are believed to be essential (Walston et al. 2010, Chapron et al. 2008, Gopal et al. 2010; 2014). As such, many extant tiger populations are by

themselves inadequate for long-term persistence (Kenney et al. 1995, Linkie et al. 2006), either because of habitats harboring a low number of breeding tigers, small size of the protected area and/ or ecologic isolation from other populations. High spatial genetic structuring and small population size observed in today's Indian tiger populations dictates preserving them in a metapopulation framework wherein individual populations are connected through a permeable habitat matrix and can occasionally exchange individuals (Hanski and Gilpin 1997). This would result in re-colonization of suitable habitat patches where tigers have become locally extinct and rescue declining local populations from extinction by immigrants (Hanski 1999, Brown and Kodric-Brown 1977). Understanding and managing the metapopulation framework of extant tiger populations is an important strategy for ensuring their long-term conservation. Connecting tiger source populations within tiger landscapes by identifying, restoring and conserving habitat corridors will not only enhance long-term tiger conservation objective but will serve to address the need for gene flow for many other biota as well.

In this document we provide a preliminary map of minimal habitat corridors connecting source populations of tigers in India. The mapped corridors are based on least cost corridors and least resistance pathways using circuit theory. Though these models are based on real life data on tiger habitat suitability obtained from extensive ground surveys during the country wide assessment of tiger status (Jhala et al 2011), however, they still need site specific ground validation for micro-adjustments. Field managers can use these as first cut information for their tiger

conservation plans that include landscape scale connectivity as an important component. The corridors shown in this report are minimal requirement. Alternative connectivities do exist in many areas and need to be conserved as well. Many of these alternative connectivities are highlighted by the resistance based connectivities (circuitscape) in this report and need conservation attention as well. The bottlenecks in the minimal corridors need special attention through enhanced protection and restoration ecology inputs. Infrastructural development within corridors should be minimal, and whenever permitted for larger National interests, it should include a "green mitigation design" for ensuring that the corridor permeability for tigers is not compromised. In the absence of any formal maps on corridor, this report will serve to provide this much needed information for long-term tiger conservation.